EXTRACTABLE SOIL PHOSPHORUS, CORRELATION WITH P FORMS IN SOIL RUNOFF, AND RELATIONSHIPS WITH THE TEXAS P INDEX AS A NUTRIENT MANAGEMENT TOOL FOR CAFOs

A Dissertation

by

FREDDY J. JACOBY

Submitted to the Office of Graduate Studies of
Texas A&M University
in partial fulfillment of the requirements for the degree of

DOCTOR OF PHILOSOPHY

May 2005

Major Subject: Soil Science

EXTRACTABLE SOIL PHOSPHORUS, CORRELATION WITH P FORMS IN SOIL RUNOFF, AND RELATIONSHIP WITH THE TEXAS P INDEX AS A NUTRIENT MANAGEMENT TOOL FOR CAFOs

A Dissertation

by

FREDDY J. JACOBY

Submitted to the Texas A&M University
in partial fulfillment of the requirements
for the degree of

DOCTOR OF PHILOSOPHY

Approved as to style and content by:

Sam E. Feagley (Chair of Committee)	Vincent A. Haby (Member)
Frank M. Hons (Member)	Mark A. Hussey (Head of Department)
Thomas W. Boutton (Member)	

May 2005

Major Subject: Soil Science

ABSTRACT

Extractable Soil Phosphorus, Correlation with P Forms in Soil Runoff, and Relationships with the Texas P Index as a Nutrient Management Tool for CAFOs. (May 2005)

Freddy J. Jacoby, B.S., California State University, Los Angeles;

M.S., Clemson University

Chair of Advisory Committee: Dr. Sam E. Feagley

Phosphorus (P) inputs into water reservoirs are the primary cause for accelerated eutrophication affecting water quality. Attempts are underway to regulate inputs originating from concentrated animal feeding operations (CAFOs). The purpose of this research was to relate runoff dissolved (DP) and total P (TP) losses to site-specific characteristics from plots in CAFOs and compare them to their corresponding risk assessment using the Texas Phosphorus Index (PI). Initial studies showed that soil test P (STP) methods used in Texas by inductively coupled plasma were highly reproducible regardless of manure source or application rate. However, NH_4OAc-EDTA extraction efficiency was increased with respect to other methods as soil conditions became less acidic, probably due to dissolution of the greater portion of Ca-bound P resulting in STP values that could be three times greater than those of Mehlich III for the same soil. Surface application of dairy manure to high pH soils were positively correlated to STP at various soil-sampling depths down to 15 cm. First order linear relationships between STP values and DP concentrations in runoff were statistically significant for extraction methods and sampling depths but were different among different soils under neutral to calcareous conditions. Attempts to reproduce this relationship on fields that received

periodic applications of manure or effluent with various incubation periods failed, although there was a single highly significant relationship between STP and runoff DP for different soils when soil conditions were acid (pH<6.5) with various sampling depths. Analyses of NH_4OAc-EDTA extractable soil elements showed Mg was significantly correlated to DP concentration across various management and soils, indicating that Mg-bound P is a major component controlling P release into runoff. Use of the Texas PI reflected vegetation type closely, with grass-covered sites averaging the lowest risk rating, and having the lowest DP and TP losses, while conversely tilled sites had the highest. However, overall relationship was poor when estimates for erosion rates were used due to experimental design limitations. Use of measured erosion rates for plots and inclusion of extractable Mg improved correlations between PI rating to DP and TP losses, with r^2 ranging from 0.60 to 0.87.

DEDICATION

This dissertation is dedicated to Humberto Jacoby, and M. Socorro Hernandez, I miss you both. To Brenda J. Burciaga, who had a different vision.

ACKNOWLEDGEMENTS

I am deeply thankful to my major advisor Dr. Sam E. Feagley for all his support, guidance, and insight throughout this investigation. Many thanks are also extended to my committee members Dr. Frank Hons and Dr. Thomas Boutton for their advice on the review of this manuscript and Dr. Vincent Haby for his patience and contributions on such short notice. Thanks to Dr. Lloyd Hossner for the use of his laboratory equipment and space. Many thanks in particular to Todd Carpenter for his overall involvement in this project and help on all those 18-hr 110° C days. Additional thanks to Frank Mazac, Scott Monk, T.J. Helton, Jim Akin, Bob Self, Donna Prochaska, Debbie Southerland and CAFO producers and the faculty and staff of the Soil and Crop Sciences department.

TABLE OF CONTENTS

CHAPTER Page

CHAPTER Page

LIST OF TABLES

TABLE Page

TABLE Page

LIST OF FIGURES

FIGURE Page

FIGURE Page

CHAPTER I

INTRODUCTION

Problem Statement

Population growth and urban sprawl are placing greater demands on lakes and streams to provide excellent quality water for drinking, recreational, industrial, and fisheries' use among others. In a 1996 report, the Environmental Protection Agency (EPA) named eutrophication as the number one factor impairing fresh water reservoirs (USEPA, 1996). Eutrophication is the natural aging of lakes and streams due to the influx of nutrients that increases biological oxygen demands and can eventually reduces oxygen (O_2) availability to other aquatic life (Pierzynski et al, 2000). Studies on lakes and streams over the last decades have shown that phosphorus (P) rather than nitrogen (N) is the most limited nutrient in fresh waters (Boyce et al., 1987; Maloney et al., 1972; Schelske and Stoemer, 1972; Schindler, 1977). Anthropogenic activities can result in unnaturally large P inputs that can increase biological activity and affect water odor, taste, and in extreme cases, cause fish kills. Legislation such as the Water Quality Act of 1972 have help to curtailed point source discharges from industry over the last few decades (Sharpley et al., 1994), yet P concentrations in streams have remained high (Correll, 1998; Smith et al., 1987).

The focus now is on nonpoint contribution, especially from agricultural land, which has been identified as the major nutrient contributor to lakes and streams (Parry,1998; USEPA, 1996; United States Geological Survey, 1999). This is partly due

---

This dissertation follows the format and style of the Journal of Environmental Quality.

to the large applications of P over the years in the form of commercial fertilizers and animal manure to attain maximum crop yields (Brady, 1990). Particular attention has been placed on areas that have high numbers of concentrated animal feeding operations (CAFOs), which has resulted in the accumulation of nutrients in soil (Lander et al., 1998). A watershed study of the North Bosque River in Texas by McFarland and Hauck (1999) was able to correlate increases of in-stream P concentrations to land area used for dairy waste application.

Because of the complexity of P movement, a soil threshold value alone is inadequate to describe the likelihood of movement into streams and lakes. While watershed-scale studies can provide a gross estimate of the nutrient contamination problem due to runoff, they fail to identify local problem areas. This is particularly important since research has shown that less than 10% of the area of a watershed is often responsible for over 90% of the P lost (Sharpley et al., 1994). The use of field-scale studies provides a logical method to allocate limited resources and target specific areas while alleviating the dependence on natural rainfall events and still giving comparable results (Kleinman et al., 2004; Sharpley and Kleinman, 2003). Significant effort has been directed toward development of tools that can estimate potential nonpoint source losses of P from fields. One example is a simple P index (PI) developed by the U.S. Department of Agriculture (USDA), Agricultural Research Service (ARS), as a field-level screening tool to rank the vulnerability of fields as sources of P loss in runoff water (Lemunyon and Gilbert, 1993). While not intended as a regulatory tool, but rather as a guideline, the PI is a better indicator than an absolute soil P value for the risk of P losses

because it includes a number of different factors and is site specific. Unfortunately initial work with the PI was limited in the soils and conditions used. Further studies on the watershed scale have shown a large variability between site ranking and actual P losses to stream (Gburek et al., 2000), indicating the need to fine-tune the PI.

Public pressure today is driving federal and state agencies to restrict nutrient amendments to land based on the potential of P contaminating water sources. However, complex interactions between P forms, cultivated crops, and land management along with limited research make regulations arbitrary at best. Because site hydrology cannot be feasibly altered, modifications to the source and application practice factors of the PI would seem to be the most practical (Gburek et al., 2000; Sharpley et al., 2000). Ongoing studies, the first of which are presented here, involved the use of field-scale rain simulations throughout the state of Texas on sites that received manure and effluent applications over a number of years. The overall goal is to use collected amounts of P (dissolved and total) in runoff to validate or modify the Texas PI as needed, so that parameters chosen and their weighted values represent a reliable method to rank fields on the likelihood of P contribution into streams and lakes.

Literature Review

Soil P

Phosphorus is an essential nutrient necessary for root development, seed formation, and plant growth (Marschner, 1995). It is also one of the most limited in

availability in soils, with most P compounds present being very insoluble and/or strongly bound onto soil particle surfaces, often resulting in plant deficiencies (Tan, 1996). Total soil P concentrations vary across soils from 50 to 1500 mg P kg^{-1} soil (Pierzynski et al., 2000), and are primarily influenced by weathering of parent material. Soils derived from acid rocks (southern Coastal Plain of U.S.) are low in P containing 0.02% to 0.04% P. Soils from basic rock (northwest region of U.S.) have high P values containing 0.09 to 0.13% P (Tan, 1996). In addition soil P is influenced by the use of fertilizers. Availability of phosphate in soils is regulated by two factors: Its low solubility and strong binding to soil particle surfaces. Most soil solutions contain less than 1 mg P L^{-1}, but can be as high as 6-8 mg P L^{-1} on recently fertilized fields (Pierzynski et al., 2000). In general, a level of 0.2 mg P L^{-1} is needed to meet nutritional needs of most plants (Wood, 1998). However, plants can only uptake P in the pentavalent forms $H_2PO_4^-$ and HPO_4^{2-}, with the former being predominant under acidic conditions and the later when soil solution pH becomes greater than 7.2. When P is first added to soils in the form of fertilizers, a rapid adsorption of P to particle surfaces takes place followed by a slower conversion into less available forms including mineral phosphates which are strongly fixed. As a result, the greatest amount of P is available to plants only during the first season after the application of fertilizers; in most cases a large portion of the remaining P is eventually released but over a number of years (Rowell, 1994). Other factors such as soil pH, solution ionic strength, and minerals present can influence the types of chemical reactions and thereby influence the solubility of P (Pierzynski et al., 2000). However,

the high P uptake by plants indicates that its availability is not simply controlled by P solubility alone (Baker and Hall, 1967).

Soil P is divided into inorganic (Pi) and organic fractions (Po). The organic P fraction values can range from 30-70% of total soil P in agricultural soils, and as high as 95% of forest soils P is organic (Harrison, 1987; Zech et al., 1987). The perception has been that most soil P is present in an organic form. However, analyses of Andisols, Aridisols and Oxisols have shown cases where the prevalent forms of P are in the inorganic fraction (Brady, 1990; Soltanpour et al 1988). Inorganic P is made up of two major mineral groups: 1) varascite-strengite group composed of $AlPO_4 \cdot 2H_2O$(varascite), $FePO_4 \cdot 2H_2O$ (strengite) and 2) apatite of which over 220 forms have been reported, including fluoroapatite ($Ca_{10}F_2(PO_4)_6$), hydroxylapatite ($Ca_{10}(OH)_2(PO_4)_6$), carbonate apatite ($Ca_{10}CO_3(PO_4)_6$), oxyapatite ($Ca_{10}O(PO_4)_6$), and chlorapatite ($Ca_{10}(Cl)_2(PO_4)_6$). The variscite-strengite fraction is associated with highly weathered acidic soils, while apatite is found in neutral to basic soils. By contrast, few generalizations can be made about the different compounds in the Po, most of which have not been identified, but whose numbers if not quantities are much greater than those in the Pi fraction. Only generally can the organic bound P compounds be separated into four groups: 1) inositol phosphates (phosphate esters), 2) nucleic acids, 3) phospholipids, and 4) miscellaneous esters.

Organic and inorganic bound P forms have different properties that can affect the amount of P in soil solutions. For example, inositol hexaphosphates, by far the most prevalent Po found in soils (10-50%), are much less mobile than other inorganic forms in

part due to their high molecular weight (Tan, 1996). In another example, while there is an accumulation of Po in acid soils, there is also a high level of Fe and Al present that usually binds Po and reduces its availability (Harrison, 1987). Since the amount of bound Po in soils is positively correlated with the organic matter content (Stevenson, 1999), a decrease in the Po fraction would be expected as organic matter decreases. This trend is most easily observed in Aridisols, where climatic conditions negatively affect bioactivity, resulting in most soil P being present in the Pi fraction (Brady and Weil, 1999). Therefore, in calcareous soils with limited rainfall and high soil pH due to the accumulation of carbonates, one would expect the majority of soil P to be inorganic rather than organically bound, which can affect the amounts of soluble P in solution.

Eutrophication

Eutrophication is defined as "an increase in the fertility status of natural water that causes accelerated growth of algae or water plants." (Pierzynski et al., 2000), and results in the subsequent decomposition of organic matter that depletes oxygen from lakes and streams. Initially, N was thought to be the most limited nutrient, but laboratory and lake experiments have proven a greater response of water plants to P influx (Boyce et al., 1987; Maloney et al., 1972; Powers et al., 1972; Rechcigl, 1995; Schelske and Stoemer, 1972; Schindler, 1977). The governing role of P in aquatic systems is reflected in its threshold which is 20-100 $\mu g\ L^{-1}$; by contrast, the N threshold is 500-1000 $\mu g\ L^{-1}$ (Pierzynski et. al, 2000). Currently, concentrations of 100 $\mu g\ P\ L^{-1}$ (USEPA, 2000) in fresh waters are considered unacceptable; however eutrophication

events have been reported with concentrations as low as 20 μg P L^{-1} (Correll, 1998). EPA guidelines recommend that P concentrations not exceed 50 μg L^{-1} in streams entering lakes, 25 μg L^{-1} in lakes, and 100 μg L^{-1} for other flowing waters not discharging directly into lakes (Eck and Stewart, 1995; USEPA, 1986). Unfortunately, excessive P levels are a nationwide problem with the average P concentration in streams being 130 μg L^{-1} (Smith et al, 1987). Although most of the P entering water reservoirs initially is believed to precipitate, anoxic conditions from seasonal changes can reduce ferric ions to ferrous and release bound phosphates at later times. Attempts to use dissolved P as an indicator for eutrophication problems have proven unsuccessful due to its fast turnover rate in the water column. In one eight-year period in the Chesapeake Bay, total P concentration increased from 20 to 200 μg L^{-1}, while simultaneously recorded values for dissolved P were only raised from 5 to 8 μg L^{-1} (Correll, 1981). Another common evaluating technique for P problems in aquatic systems is the use of N:P ratio (Redfield, 1958). Unfortunately, this method is both time limited (affected by storms) and fails to consider long-term mineralization of particulate matter. These factors prevent accurately assessing the condition of reservoirs until algal blooms occur. Therefore, in order to mitigate eutrophication of lakes, efforts must be geared to monitor P losses from fields before problems occur.

Transportation of P

Total P losses from soils depend on transport (rainfall, irrigation, erosion, infiltration and runoff), source (P forms, soil content), and management factors

(application method, timing etc.). Phosphorus can be transported by runoff and leaching from agricultural fields into lakes and streams, either in its soluble form (dissolved) or as sediment-bound P (minerals and organic matter). Except in sandy coarse-textured or organic peaty soils, P losses to groundwater through leaching are not significant (Sims et al., 1998). By far the greatest P losses are due to runoff, but which forms are being removed is still not clear. Analyses of runoff from cultivated fields indicate that particulate bound P is the greatest portion lost, with inorganic soluble P making up 10 to 40% of transported P (Sharpley et al., 1992). There is also evidence that in manure amended fields as much as 50% of the transported P can be as soluble organic bound P (Pierzynski et al., 2000), which complicates the prediction P losses impact on surface water.

Factors that can influence P transport include soil properties, types of fertilizer used, and farming practices. In the cases of morphological characteristics, sandy soils that are not strongly acidic and those where inorganic colloids are absent retain very little of the P added as fertilizer; however, as soils become finer in texture, P retention increases (Fox and Kamprath, 1971; Harris et al, 1994; Sonzogoni et al, 1980). This can lead to the contamination of water through leaching of P if large applications of fertilizers are applied to sandy sites with little clay content. Though total P transported through leaching is small with respect to runoff, as much as 13% of added P from manure has been documented to move down 60 to 90 cm (Vetters and Steffens, 1980). A possible reason is that the large amount of organic compounds helps to chelate Al and

Fe present, thereby allowing P to move down the profile before it precipitates or it is bound.

Another theory is that the organic compounds coat colloidal surfaces responsible for P adsorption, allowing for more rapid and deeper leaching from manure derived P (Harris et al., 1996). Similar effects have been observed in calcareous soil, where organic materials increased the solubility of P more than the addition of inorganic P alone (O'Connor et al., 1986). This behavior has been explained by showing that, in calcareous soils, organic acids increase the solubility of dicalcium phosphate dihydrate - a precursor for hydroxyapatite, and the major calcium bound P-form at high pH values (Grossl and Inskeep, 1991; Inskeep and Silvertooth, 1988). Therefore, the likelihood for shallow water table contamination would seem to be greater through the use of manures than inorganic fertilizers.

Land use can also affects how much P is lost from fields since mobility is greater in arable cropping systems primarily due to tilling practices than in low-intensity grassland. Using no-till can reduce runoff losses by as much as 81% (Andraski et al., 1985). Management also can alter the Pi to Po ratio in soil. Continuously cropped systems can result in declines of total soil P by about 29% compared to pastures, with these losses being greatest for the stable (residual) P forms followed by the extractable inorganic and organic P at 52, 26, and 22% of total P lost, respectively (Hedley et al., 1982). The use of NO_3 fertilizer and high temperatures also has been reported to reduce water soluble P (Kuo and Jellum, 1987). This is particularly important because soluble P levels are low during spring/summer and increase to the maximum values in winter,

when the first high-intensity rainfall event can produce the highest P losses in the runoff (Edwards and Daniel, 1993).

Determination of Soil P Fractions and P Availability Indices

Determination of total P in soils has commonly involved the complete oxidation of all forms into soluble orthophosphate through either chemical digestion with a strong acid (Mehta et al., 1954; Olsen and Sommers 1982; Tan 1996), or by ignition with Na_2CO_3 (Saunders and Williams, 1955). Subsequently, soluble P is measured colorimetrically by a reaction with molybdate (Murphy and Riley, 1962) or the use of inductively coupled plasma (ICP) analysis. Unfortunately, problems for both methods arise when attempting to quantify the Pi and Po fractions. This is because Po is determined indirectly by evaluating the Pi fraction and subtracting it from the total soil P. In many cases, during Pi determination unintended hydrolysis of the organic P ester compounds by acid takes place, thereby altering the distribution of P forms. In the ignition method, high temperatures alter the solubility of the inorganic fraction, resulting in an overestimation of Po, especially in mineral and highly weathered soils (Harrison, 1987; Kuo, 1996; Williams et al., 1970). Other factors interfering with accurate determinations of P forms in soils include hydrolysis of organic bonds by enzymes that are released during the drying and grinding of soils. The end result can be a significant over or underestimate of Pi, which has been shown to be the only form immediately available for aquatic plant uptake (Peters, 1981) and hence might be the best indicator of bioavailable risk.

In the case of P availability index, various soil extraction methods have long been used to access the status of soils in order to make crop recommendations. The most used soil test P (STP) methods in the U.S. can be classified into four main types: 1) Water or unbuffered salt solutions, 2) Dilute weak acid solutions with or without complexing agents 3) Dilute strong acid solutions with or without complexing agents, and, 4) Buffered alkaline solutions. Even though studies have shown a correlation between STP values and extractable and desorbable P in the runoff from soils with high P contents (Edwards and Daniel 1993; Pote et al. 1996, 1999; Sharpley, 1995a), these tests were designed to correlate extracted P from inorganic fertilizers to crop yields, not to predict runoff P and its effect on aquatic plant nutrient uptake and water quality. While highly reproducible and expedient, STP values do not provide information for soluble organic P concentrations (colorimetric techniques), mineralization rates, or give an indication whether the various forms of P might become sources of nutrient in the future. Interpreting STP values is further complicated because of the different chemicals used, extracting periods, and ratios of soil to extracting solution (Jones, 1998, Sharpley et al., 1994). Some of these methods were designed to be used with low soil pH values (Mehlich, 1984; Bray and Kurtz, 1945), while others are better suited for calcareous soils (Ashworth and Mrazek, 1995; Olsen et al., 1954; Olsen and Sommers, 1982). In Texas, because of its diverse climate and soils, an acidified ammonium acetate-EDTA extraction, referred to here as the TAMU extract (changed to Mehlich III, effective 20 Jan. 2004), had been used for years by the Texas Cooperative Extension (TCE) Soil, Water and Forage Testing Laboratory as a multiple fertility test (Texas A&M Univ.

Agric. Ext. Serv. 1980) because it correlated well to various extraction methods in predicting crop yields (Hons et al., 1990).

Even when the same extractant is used, there is a large degree of variability in extraction times and ratios used among labs. This is particularly true with water (Indiati and Sharpley, 1997; Kuo, 1996; Olsen and Sommers, 1982; Olsen and Wantanabe, 1970; Van der Paauw, 1971; Van Diest, 1963) and $CaCl_2$ (Labhsetwar and Soltanour, 1985; Soltanour et al., 1974; White and Beckett, 1964; Dalal and Hallsworth, 1977) extractions which have been shown to correlate the best with available P found in runoff waters (Edwards and Daniel, 1993; Pote et al., 1996). Furthermore, many labs still measure P using colorimetric methods, which do not include the soluble organic P forms, that, in theory, should and usually results in lower P concentrations than when ICP techniques are used (Labhsetwar and Soltanpour, 1985). The lack of a consistent protocol along with hydrological and topographical differences makes relying on STP to assess risk of P contamination difficult. In addition no calibration between P losses and STP values on a large scale has been conducted.

CAFOs and Animal Manure Properties

Prior to the use of commercial fertilizers, manure was a highly valuable source of soil nutrients. However, long-term applications have led to elevated soil P levels especially in areas with large numbers of livestock operations (Sims, 1993). The increase in soil P levels is due to past practices, such as manure applications based on crop N requirements. For instance, most fresh manures have N:P ratios of 3 or 4 to 1, and because of N volatilization are usually applied at ratios less than 3:1. The overall

result is the use of P amounts three to five times greater than plant requirements (Eck and Stewart, 1995) because most plants contain N:P ratios of 6 or 7 to 1. In Texas, the application of manure from CAFOs is stringently regulated, with nutrient management based on N until a 200 mg kg^{-1} (200 ppm) extractable soil test P threshold value is obtained. After reaching this soil test P threshold, additions are P based relative to crop requirement. Another effect of the proliferation of CAFOs in certain regions of the U.S. has been the fragmentation between areas where feed is produced and livestock are raised (Lanyo and Thompson, 1996). A study by the National Research Council (1993) estimated that only 30% of imported fertilizer feed P is exported from CAFOs as crop or animal products. Because of transportation costs, large quantities of manures enriched in nutrients as a result of dietary feed, have been spread over relatively small areas for a number of years raising all forms of soil P (Pyerzynski et al., 2000; Sharpley et al., 1984) and leading to a greater risk that nutrient losses can occur during storm events through runoff and erosion processes. Something that various scientists have already confirmed by showing a clear correlation between increases in total soil P and amounts of P released in water runoff as the result of manure application (Edwards and Daniel, 1993; Pote et al., 1996, 1999; Sharpley et al., 1984, 1992; Sharpley, 1993, 1995a; Shreve et al., 1995). While cattle and swine operations have been under this plan for some time, dry-poultry facilities had been exempted when this project was started. However, as of Apr. 2003, all animal operations are now in the same EPA regulations.

The P content found in manure is dependent on source, diet, and age. In general poultry broilers generate manures with the highest total P content, followed by laying

hens, swine slurry, beef cattle, and dairy at 1.8, 1.2, 0.34-2.45%, 0.22-0.90, and 0.6% P, respectively (Destain, 1980; Larsen, 1980; Pierzynski et al., 2000; Whithers and Sharpley 1995), of which the inorganic fraction is 91, 84 and 63% for swine, poultry, and dairy, respectively (Sharpley and Moyer, 2000). This is in partly due to the animals' different abilities to extract P. Laying hens, swine, cattle, and broilers excrete 91, 77, 70, and 61% respectively, of the total P in feed. The use of supplement feed can also alter P uptake. For example, the inclusion of calcium phytate in feed, commonly used in cattle feed, was shown to decrease P absorption by 60 to 75% when given to monogastric animals (Simons and Jongbloed, 1980). However, the addition of the phytase enzyme, along with phytic acid to poultry has been shown to reduce total P concentration of the resulting manure while increasing the amount of water extractable P (Vadas et al., 2004). This can lead to greater losses of dissolved and total P in runoff when manure is applied on a total P basis because of the higher application rates of manure which are then more susceptible to erosion (Penn et al., 2004). Another factor to consider is the age of the manure being surface applied. Fresh manures initially have higher Po contents that decrease overtime as the C:P ratio falls below 200 (Pierzynski et al., 2000). Indications are that the Pi fraction eventually accounts for 68 to 81% of the total available P (Sharpley et al., 1984; Sims, 1992) with complete turnover of dissolved Pi through microorganisms and decomposition estimated to take 10 to 20 weeks (Gerritse, 1980). While manure amendments can increase overall soil P content (Gartley and Sims, 1994), it is unclear whether it is due to the organic-bound fraction, or just a general increase (Harrison, 1987; Webber, 1980). In an eight-year trial, Sharpley et al.

(1984) recorded increases in total, inorganic, and available P, with Pi increasing the most (increasing from 34 to 71%). However, after cessation of applications, Po returned to previous levels faster than did other forms. This could indicate that long-term manure addition can result in higher mineralization rates while maintaining higher levels of the soluble Pi fraction.

An interesting note to consider is that while total P accumulates through the use of sewage sludge or animal manure amendment, extractable P and its leaching potential have been shown to be different. In general, STP (30-70%) from manure amended sites tend to be higher than those with sewage sludge (0.8-6%), even with P contents that are higher in sewage sludge (Johnston, 1980; Wen et al., 1997). When excessive amounts of sludge and manure were used, P leaching to significant depths was only reported for sites that had manure applied (Suss, 1980; Furrer, 1980). The explanation for the solubility difference is believed to be due to the greater quantities of Fe and Al present in sludge that can caused the precipitation and adsorption of P. This is supported by an observed decreases of P in runoff from fields that had poultry litter applied in conjunction with either alum ($Al_2(SO_4)_3 \cdot 14H_2O$) or ferrous sulfate ($FeSO_4 \cdot 7H_2O$). When rainfall simulations were conducted in this plots 2 and 16 days after application, alum reduced runoff P by 87 and 63%, respectively (Shreve et al., 1995). Ferrous sulfate decreased P loss by 77 and 48% in those same timed periods. The overall effect is that P is more mobile in the soil profile in manure-treated sites and hence has a higher potential to contaminate shallow water tables.

Soil P Index

The use of a nationwide soil P threshold value has been proposed, but as has been mentioned earlier, such values are soil-specific and research would have to be conducted for all soil types. A more practical method is to integrate STP with other factors to assess risk. One proposed tool to balance levels necessary for maximum crop yields against likely P losses transported to aquatic environments is through the use of a Phosphorus index (PI) first proposed by Lemunyon and Gilbert (1993). Developed by the U.S. Department of Agriculture, Agricultural Research Service, and several research scientists, the P index is a helpful tool used to evaluate and rank the relative risk that P might reach surface water by taking into account site-specific factors that influence P movement (i.e. slope, rain patterns, distance to surface water etc.) in addition to application rates, source, and timing. The P index was first published for NRCS use in the South National Technical Center publication, Engineering Technical Note 1901, A Phosphorus Assessment Tool, August 1994. Because of the large area and variety of weather conditions found in Texas, the Phosphorus Assessment Tool (USDA/NRCS, 2000a & b) or PI has been divided into two regions: East and West Texas based on an annual precipitation basis. The current model uses an 8 x 5 matrix, where eight site characteristics are collected and then rated on a five-value qualitative scale (very low, low, medium, high, and very high). Weighting factors include Soil Test P Level (1.0), Fertilizer P Application Rate (0.75), Organic P Application Rate (0.75), Fertilizer P Application Method and Timing (0.50), Organic P Source/Application Method and Timing (0.50), Proximity of Field to Edge of a Stream or Lake (1.25), Runoff Class (1.0)

and Soil Erosion (1.50). Most of the values, scales, and ratings used in the PI were based on scientific judgment for source and transport factors to rate risk.

Objectives

The primary goal of this project was to determine if the use of the current Texas PI accurately reflected P losses from various fields with different physical properties and management practices. We first determined the reproducibility of STP values with seven different extraction methods using ICP techniques on soils collected throughout the Central and Eastern portion of the U.S. that have different animal manures added. Previous studies have found a correlation between dissolved P in runoff and STP values, but in relatively small areas and mostly in conditions of acid soils. After choosing three of the most reliable P extraction methods, we tested the relationships between STP and runoff dissolved P with different loading rates of dairy manure on four common soil series in the North Bosque River area in Central Texas. Finally, twenty-three feedlot and dairy CAFOs located in the Southern High Plains and Central Texas regions were used for rainfall simulation studies. Total P losses were used to evaluate whether the current Texas PI is a valid nutrient management tool in assessing the risk of P loss from a specific site based on the current parameters being used.

CHAPTER II

PHOSPHORUS EXTRACTION METHODS EVALUATION FROM ANIMAL MANURE APPLICATION FIELDS

Synopsis

Soil testing has been used over the years to predict crop responses to fertilizer addition, and varies widely across regions due to climatic and soil differences. Today, there are attempts to use soil test values as indicators of environmental quality. However, field correlations between extractable nutrients and their effects on the environment have not been established. Establishing a correlation is further complicated by different methods used. In the case of P, soil pH plays a key role in the forms present in soils and their solubility. Most studies to date have used colorimetric techniques to measure soluble P, which measure only inorganic P that reacts with molybdate (Olsen and Sommer, 1982). We were interested in determining if there was a difference in reproducibility when inductively couple plasma spectrometry (ICP) was used to measure the total soluble P using different extracts on soils with different manure sources.

Materials and Methods

Soil Collection and Preparation for Soil Test Phosphorus Studies

At least 45.4 kg of surface soil (0 to 15 cm) were collected from sites that had received manure applications from seven different states (Table 1). These soils varied in texture, soil pH, and source of animal manure (poultry, swine, and beef and dairy cattle)

that had been applied. Soil series name, texture, classification, pH and location are listed in Table 1. All soils had received large applications (2-30 t/Ac/yr) poultry, swine effluent, dairy or feedlot manure over several years (longest 50 years). Samples were air dried for two to three weeks and then pulverized using a Dinocrush grinder. Debris were removed using a 2-mm sieve and soils were then stored in plastic containers at room temperature. Soil pH was measured using a 1:2 ratio (15.0 g of soil and 30 mL deionized water). Samples were thoroughly stirred and allowed to equilibrate at room temperature for at least 30 minutes, but no more than 4 hours (Provin 2003; Texas A&M University Agricultural Extension Service, 1980). The soil-water mixture was once again briefly stirred prior to the insertion of the pH electrode. Readings were taken using a Corning pH meter 145 after 2 minutes. In order to prevent the effects of junction potential errors care was taken not to immerse the electrode into the solid phase of the mixture. Calcium carbonate analysis was performed using a Chittick apparatus only on soils that effervesced when 1N HCl was added and are presented in Table 2 (Dremanis, 1962).

Soil Phosphorus Extraction Procedures

Extractable P on each soil was determined using the following methods: Bray I (Bray and Kurtz, 1945), Mehlich III (Mehlich, 1984), Olsen (Olsen et al., 1954; Olsen and Sommers, 1982), acidic ammonium acetate-EDTA (Hons et al. 1990; Texas A&M University Agricultural Extension Service, 1980), as well as deionized water and the dilute solutions $CaCl_2$ and KCl (Kuo, 1996; Soltanpour et al., 1974; Tan, 1996). The

Table 1. Soil series and manure source information.

Soil Series	Location	Type of manure	Soil pH	Texture	Soil Classification
Portneuf	ID	Dairy	8.42	Silt Loam	Coarse-silty, mixed, superactive, mesic Durinodic Xeric Haplocalcids
Purves	TX	Dairy	8.37	Clay	Clayey, smectitic, thermic Lithic Calciustolls
Blanket	TX	Dairy	8.31	Clay Loam	Fine, mixed, thermic Pachic Argiustolls
Houston Black	TX	Dairy	7.84	Clay	Fine, Smectitic, thermic Udic Haplusterts
Windthorst	TX	Dairy	7.52	Fine Sandy Loam	Fine, mixed, thermic Udic Paleustalfs
Crockett	TX	Dairy	6.94	Fine Sandy Loam	Fine, smectitic, thermic Udertic Paleustalfs
Berk	PA	Dairy / Poultry	6.56	Loam	Loamy-skeletal, mixed, active, mesic Typic Dystrudepts
Watson	PA	Dairy	6.33	Silt Loam / Silty Clay Loam	Fine-loamy, mixed, mesic Typic Fragiudults
Perico	TX	Feedlot	7.97	Fine Sandy Loam	Fine-loamy, mixed, mesic Aridic Paleustalfs
Richfield	OK	Feedlot	7.96	Silt Loam / Silty Clay Loam	Fine, smectitic, mesic Aridic Argiustolls
Captina	AR	Poultry	6.50	Silt Loam	Fine-silty, siliceous, active, mesic Typic Fragiudults
Ruston	LA	Poultry	5.98	Fine Sandy Loam	Fine-loamy, siliceous, semiactive, thermic Typic Paleudults
Hartsells	AL	Poultry	5.82	Fine Sandy Loam	Fine-loamy, siliceous, thermic Typic Hapludults
Ruston	LA	Poultry	5.54	Fine Sandy Loam	Fine-loamy, siliceous, semiactive, thermic Typic Paleudults
Ruston	LA	Poultry	5.52	Fine Sandy Loam	Fine-loamy, siliceous, semiactive, thermic Typic Paleudults
Cuthbert	TX	Poultry	5.38	Fine Sandy Loam	Fine, mixed, semiactive, thermic Typic Hapludults
Portneuf	ID	Swine	8.46	Silt Loam	Coarse-silty, mixed, superactive, mesic Durinodic Xeric Haplocalcids
Richfield	OK	Swine	7.61	Silt Loam / Silty Clay Loam	Fine, smectitic, mesic Aridic Argiustolls
Dallam	TX	Swine	6.86	Fine Sandy Loam / Sandy Clay Loam	Fine-loamy, mixed, mesic Aridic Paleustalfs
Captina	AR	Swine	6.30	Silt Loam	Fine-silty, siliceous, active, mesic Typic Fragiudults
Captina	AR	Swine	5.99	Silt Loam	Fine-silty, siliceous, active, mesic Typic Fragiudults

Table 2. Calcium carbonate equivalent (CCE) of 21 soils with manure applications.

Soil Series	Soil pH	Manure Source	Calcite	Dolomite	CCE
			g / kg of soil		
Cuthbert	5.38	Poultry	†	†	†
Ruston	5.52	Poultry	†	†	†
Ruston	5.54	Poultry	†	†	†
Hartsells	5.82	Poultry	†	†	†
Ruston	5.98	Poultry	†	†	†
Captina	5.99	Swine	†	†	†
Captina	6.30	Swine	†	†	†
Watson	6.33	Dairy	†	†	†
Captina	6.50	Poultry	†	†	†
Berk	6.56	Dairy / Poultry	†	†	†
Dallam	6.86	Swine	†	†	†
Crockett	6.94	Dairy	†	†	†
Windthorst	7.52	Dairy	†	†	†
Richfield	7.61	Swine	†	†	†
Houston Black	7.84	Dairy	†	†	†
Blanket	8.31	Dairy	†	†	†
Purves	8.37	Dairy	†	†	†
Perico	7.97	Feed yard	120	10	130
Portneuf	8.42	Dairy	440	290	750
Richfield	7.96	Feed yard	760	150	920
Portneuf	8.46	Swine	920	390	1340

† Soils did not effervesce when 1N HCl was added.

first four tests are currently used as fertility tests in the U.S. While extractable P in water and dilute salts has been proposed as a good predictor of dissolved P in runoff, there is not a common protocol, with extracting ratios ranging from 1:1.25 to 1:100

(Indiati and Sharpley, 1997; Olsen and Sommers, 1982; Olsen and Wantanabe, 1970; Van Diest, 1963; Van der Paauw, 1971), and reaction shaking times from 5 min up to 15 hrs (Kuo, 1996; Olsen and Sommers, 1982; Van Diest, 1963). Therefore, we performed a series of experiments to determine soil to solution ratios, shaking-times, and salt concentration that would give the most reproducible P concentrations (Data not shown). A ratio of 1:5 (5.0 g soil: 25 mL water) and 5 minutes shaking was chosen based upon a lower coefficient of variation and to minimize the volume that needed to be filtered, since filtration (2-μm filter) of water solution could take up to four hours even after centrifugation. Different concentrations, soil to extractant ratios, and shaking times were also tried with $CaCl_2$ and KCl (Data not shown). Based upon reproducibility, a 1:5 dilution, 5-minute shaking time, and $CaCl_2$ and KCl concentrations of 0.1 M and 1 M, respectively, were chosen. All extractions and subsequent filtrations were performed at room temperature (25^{o} C). Samples were then stored at 4^{o} C if inductively coupled plasma (ICP) analysis could not be performed within an hour.

Statistical Analysis

Soil extractable P means and standard deviations for each method were calculated using 10 replicates. Reproducibility was assessed by comparing coefficient of variability (CV) across different manure sources and soil pH range. Pair-wise differences among means for each method within each soil series were made using a simple t-test of independent samples. All analyses follow standard format using SPSS software version 11.01 (SPSS, 2001).

Results and Discussion

Extracting Efficiency

The soils used had a broad range of physical and chemical properties, with soil pH ranging from 5.38 to 8.42, and texture from fine sandy loam to clay. Overall, poultry applications were in the slightly acidic soils range while those of beef and dairy cattle were applied to soils of a calcareous nature. This was partly due to location and dietary feeds of these animals. Swine manure was found on soils of both pH ranges. The amounts of extractable P for all extraction methods are listed in Tables 3 and 4. The methods can be divided into two groups based on their extracting abilities: deionized water and salt extractions that yielded the lowest amounts of labile P, and conventional soil fertility tests which had the highest P concentration values. Neither the source of animal manure, nor the time that soils received manure affected extracting efficiencies of the different methods with respect to each other.

Water and Salt Extracting Solutions

A clear difference between extractable P concentrations from water and dilute salt extractions ($p < 0.01$) was obtained. Water solutions consistently yielded the highest P concentrations ranging from 3.1 up to 63.4 mg P kg^{-1} soil, in all but one case. One soil had 128.0 mg P kg^{-1} soil for water STP, however, this sample (Hartsells) was collected by a third party and appeared to have been composed primarily of poultry litter residue rather than soil, which would account for the extremely high levels of extractable P that were obtained regardless of method used, however the order of P extraction efficiency

Table 3. Comparison of soil phosphorus extracting methods among 10 soils with dairy or feedlot manure from 7 states.

DAIRY MANURE

Soil	Soil	$CaCl_2$			KCl			Water			TAMU			Bray I			Olsen			Mehlich III		
Series	pH	mg/kg	SD	CV	mg/kg	SD	CV	mg/kg	SD	CV	mg/kg	SD	CV	mg/kg	SD	CV	mg/kg	SD	CV	mg/kg	SD	CV
Watson	6.33	2	0.1	5.4	3.4	0.1	4.0	3.1	0.3	10.7	130.3	4.3	3.3	88.5	1.4	1.6	102.1	5.3	5.2	177.9	2.2	1.2
Berk	6.56	5.3	0.1	2.7	10.3	0.3	2.8	17	1.1	6.2	264.7	12.6	4.7	324.9	5.1	1.6	202.4	7.6	3.8	530.1	10.5	2.0
Crockett	6.94	2.9	0.3	9.8	8.3	0.3	4.1	11.9	0.4	3.1	292.8	15.8	5.4	235.4	4.6	2.0	165.8	7.2	4.3	409.2	9.2	2.2

Soil	Soil	$CaCl_2$			KCl			Water			TAMU			Bray I			Olsen			Mehlich III		
Series	pH	mg/kg	SD	CV	mg/kg	SD	CV	mg/kg	SD	CV	mg/kg	SD	CV	mg/kg	SD	CV	mg/kg	SD	CV	mg/kg	SD	CV
Windthorst	7.52	16.1	-	-	7.5	-	-	11.1	-	-	240	-	-	223.4	-	-	175.5	-	-	270.3	6.8	2.5
Houston Black	7.84	1.3	-	-	3.2	-	-	4.8	-	-	216.8	11.7	5.4	12	2.4	20.0	169.1	6	3.6	147	8.1	5.5
Blanket	8.31	3.8	-	-	2.8	-	-	4	-	-	120	-	-	4.4	-	-	64.5	-	-	97.1	5.3	5.5
Purves	8.37	1.2	-	-	2.8	-	-	3.1	-	-	189	-	-	4.7	-	-	127.8	-	-	54.9	6.7	12.2
Portneuf	8.42	2.9	0.1	3.4	5.4	0.1	1.9	11.3	0.4	3.5	234	3.8	1.6	35.9	4.1	11.4	104.5	7.4	7.0	147.2	6.4	4.4

FEEDLOT MANURE

Soil	Soil	$CaCl_2$			KCl			Water			TAMU			Bray I			Olsen			Mehlich III		
Series	pH	mg/kg	SD	CV	mg/kg	SD	CV	mg/kg	SD	CV	mg/kg	SD	CV	mg/kg	SD	CV	mg/kg	SD	CV	mg/kg	SD	CV
Richfield	7.96	2.7	0.2	5.3	6.2	0.2	3.6	10.2	0.3	3.3	657.4	17.9	2.7	49.7	11.1	22.4	105	5.3	5.1	266	3.7	1.4
Perico	7.97	13.8	0.2	1.4	42.8	0.8	1.9	63.4	2	3.2	887.5	33.9	3.8	423.5	9.2	2.2	363.2	7.5	2.1	679.9	15.5	2.3

Table 4. Comparison of soil phosphorus extracting methods among 11 soils with swine effluent or poultry manure from 7 states.

SWINE EFFLUENT

Soil	Soil	$CaCl_2$			KCl			Water			TAMU			Bray I			Olsen			Mehlich III		
Series	pH	mg/kg	SD	CV	mg/kg	SD	CV	mg/kg	SD	CV	mg/kg	SD	CV	mg/kg	SD	CV	mg/kg	SD	CV	mg/kg	SD	CV
Captina	5.99	4.8	0.2	4.5	5	0.3	5.1	15.8	0.3	2.1	72.6	4	5.5	80.7	2.2	2.7	97.8	4.1	4.2	150.4	2.4	1.6
Captina	6.30	3.6	0.1	2.4	3.8	0.1	2.4	14.5	1.3	9.2	88.8	4.7	5.3	89.8	2.6	2.8	92.59	2.5	2.7	160.5	5.4	3.4
Dallam	6.86	8.4	0.3	3.6	8.3	1.2	14.4	22.9	1	4.5	105.4	6	5.7	120	1.7	1.4	73.9	2.2	3.0	146.2	1.4	0.9

Soil	Soil	$CaCl_2$			KCl			Water			TAMU			Bray I			Olsen			Mehlich III		
Series	pH	mg/kg	SD	CV	mg/kg	SD	CV	mg/kg	SD	CV	mg/kg	SD	CV	mg/kg	SD	CV	mg/kg	SD	CV	mg/kg	SD	CV
Richfield	7.61	2.1	0.2	10.5	2.5	0.4	16.6	9.4	1.6	17.3	184.3	4.8	2.6	83.6	3.3	3.9	47.9	3.7	7.8	173.1	10.8	6.2
Portneuf	8.46	3	0.1	2.7	8.3	0.3	3.5	14	1.2	8.3	307.8	15.6	5.1	16.6	5.2	31.4	164.5	7.9	4.8	193.2	2.9	1.5

POULTRY MANURE

Soil	Soil	$CaCl_2$			KCl			Water			TAMU			Bray I			Olsen			Mehlich III		
Series	pH	mg/kg	SD	CV	mg/kg	SD	CV	mg/kg	SD	CV	mg/kg	SD	CV	mg/kg	SD	CV	mg/kg	SD	CV	mg/kg	SD	CV
Cuthbert	5.38	5.3	0.3	5.9	6.7	1.1	15.7	17.9	0.6	3.4	102.9	14.4	14.0	150.5	3	2	115.7	2.4	2.0	229.4	8.1	3.5
Ruston	5.52	4.2	0.1	2.3	3.8	0.1	3.4	13.6	0.8	6.1	48.5	2.3	4.8	145.5	3.8	2.6	97.7	4.9	5.0	198.9	5.1	2.6
Ruston	5.54	2.3	0.1	2.6	1.7	0.1	4.9	9.7	0.4	4.5	44.3	2.1	4.7	135.6	4.1	3	119.7	7.7	6.4	210.3	8.4	4.0
Hartsells	5.82	66.9	1.5	2.2	91	2.2	2.4	128	3.7	2.9	735.9	19.5	2.7	369.2	10	2.7	429.7	11.8	2.7	744.6	23.1	3.1
Ruston	5.98	3.4	0.1	2.6	2.9	0.1	3.4	13.6	0.8	5.5	58.04	4.6	7.9	149.5	0.5	0.3	116.7	3.2	2.8	227.1	2.6	1.1
Captina	6.50	11.5	0.3	3.0	16.4	0.3	1.9	42.3	2.3	5.4	788.9	30.2	3.8	408.6	9.2	2.2	319.6	25.2	7.9	891.6	18.8	2.1
Berk	6.56	5.3	0.1	2.7	10.3	0.3	2.8	17	1.1	6.2	264.7	12.6	4.7	324.9	5.1	1.6	202.4	7.6	3.8	530.1	10.5	2.0

* The Hartsells series was collected by a third party and appears to be made of mostly poultry litter residue rather than soil.

among extractants remained the same as in all other soils. The KCl salt extraction procedure had a lower extracting efficiency (1.7 – 42.8 mg P kg^{-1} soil) than water but consistently higher than the less concentrated $CaCl_2$ (1.3 – 16.1 mg P kg^{-1} soil) for all treatments. Figure 1 shows extractable values for all three-extraction solutions from sites with different animal manure sources.

The use of either calcium or potassium chloride gave a clear filtrate as opposed to deionized water; however reproducibility by ICP analysis was good for all three methods with all twenty-one soils having CV's consistently below 10. The use of water extractable P has been proposed to better reflect available P under natural conditions and one of the best correlated to dissolved P (DP) found in runoff (Pote et al., 1996). Unfortunately, it has been one of the most cumbersome methods. Even following centrifugation of extracting solutions, filtration of 25 mL could exceed eight hours. The large amounts of suspended particulate matter, especially when fresh manure is present, will preclude this procedure from being adopted by most laboratories doing routine analysis. In order to compare extracting efficiencies, we divided extractable P for each method by the lowest extracted concentrations. In all cases, the lowest quantity of P was extracted by $CaCl_2$ (Table 5). Generally, extractable P for water and KCl procedures with respect to $CaCl_2$, increased as soil pH increased.

Extracting Procedures Normally Used for Evaluation of Soil Fertility

Initially, at low soil pH values (below 6.86), P extraction for the four soil fertility extraction procedures, was in order from high to low: Mehlich III, Bray I, Olsen, and TAMU regardless of manure source (Fig. 2). As soil pH increased, indicative of

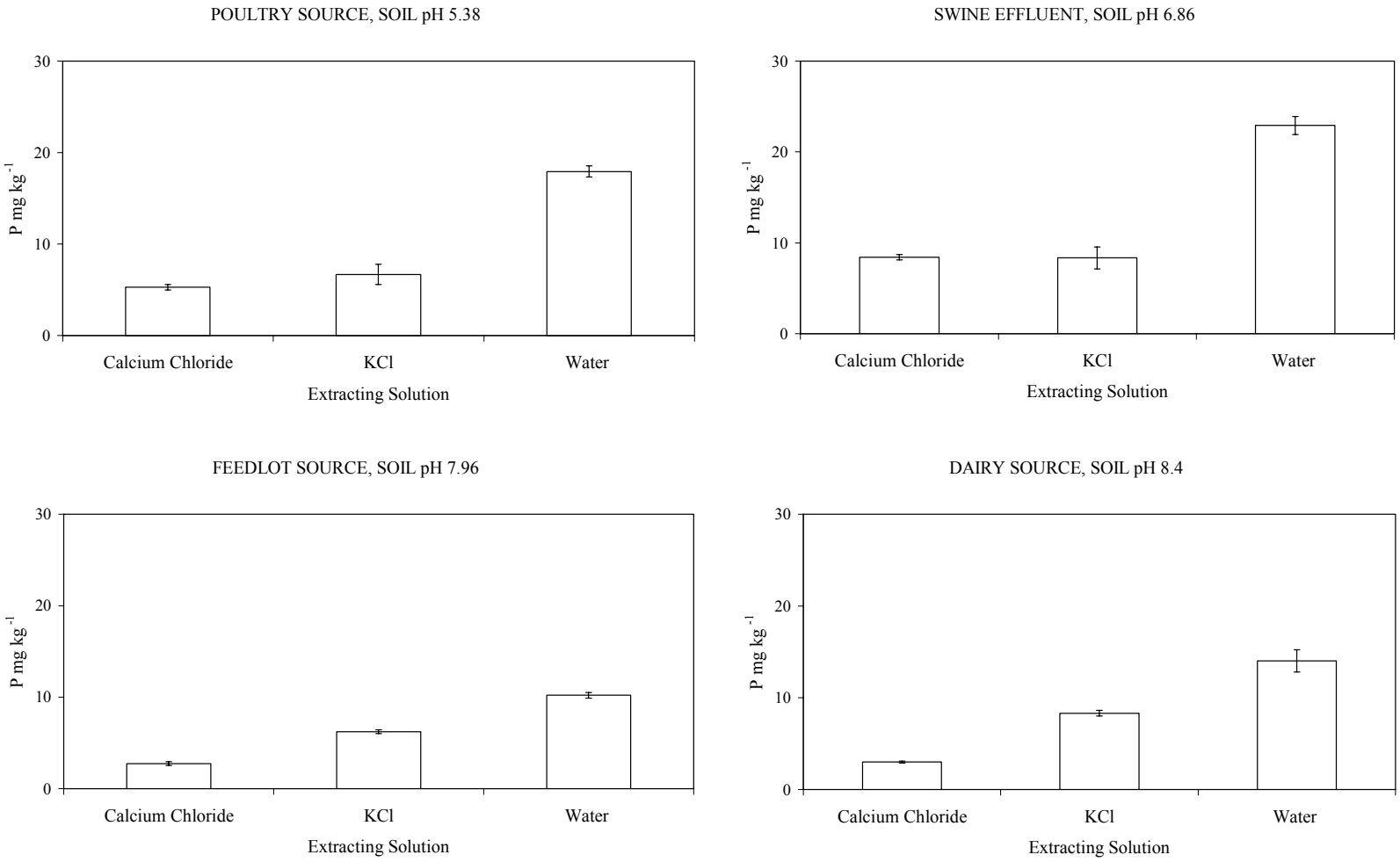

Fig. 1. Effects of manure source on extraction of P by $CaCl_2$, KCl and water.

Table 5. Extractable phosphorus ratios with respect to calcium chloride, standard deviation (SD), and coefficient of determination (CV) for the pH range.

pH Range		$CaCl_2$†	KCl‡	Water §	TAMU¶	Bray I#	Olsen††	Mehlich III‡‡
5.0 - 8.5	Mean	1	1.90	3.70	64.90	27.90	38.50	62.30
	SD		1.40	1.30	61.40	21.50	30.10	34.70
	CV		75	35	95	77	78	56
	r^2		0.32	0.19	0.39	0.21	0.12	<0.01
5.0 - 6.0	Mean	1	1.00	3.30	15.50	31.20	26.20	48.30
	SD		0.20	0.80	3.70	18.90	15.30	27.70
	CV		24	24	24	61	58	57
	r^2		<0.01	0.02	0.09	0.10	0.04	0.07
6.0 - 7.5	Mean	1	1.60	3.10	48.40	39.50	31.60	69.90
	SD		0.60	1.00	32.80	25.00	18.80	46.00
	CV		39	32	68	63	59	66
	r^2		0.02	0.05	0.07	0.05	0.17	0.11
7.5 - 8.5	Mean	1	2.70	4.50	116.40	15.20	53.80	66.10
	SD		1.90	1.40	66.70	14.00	41.00	29.00
	CV		70	32	57	92	76	44
	r^2		0.23	0.07	0.05	0.59	<0.01	0.44

† $CaCl_2$, calcium chloride solution extracted P.
‡ KCl, potassium chloride solution extracted P.
§ Water, deionized water extracted P.
¶ TAMU, ammonium acetate-EDTA extracted P.
# Bray I extracted P
†† Olsen extracted P
‡‡ Mehlich III extracted P

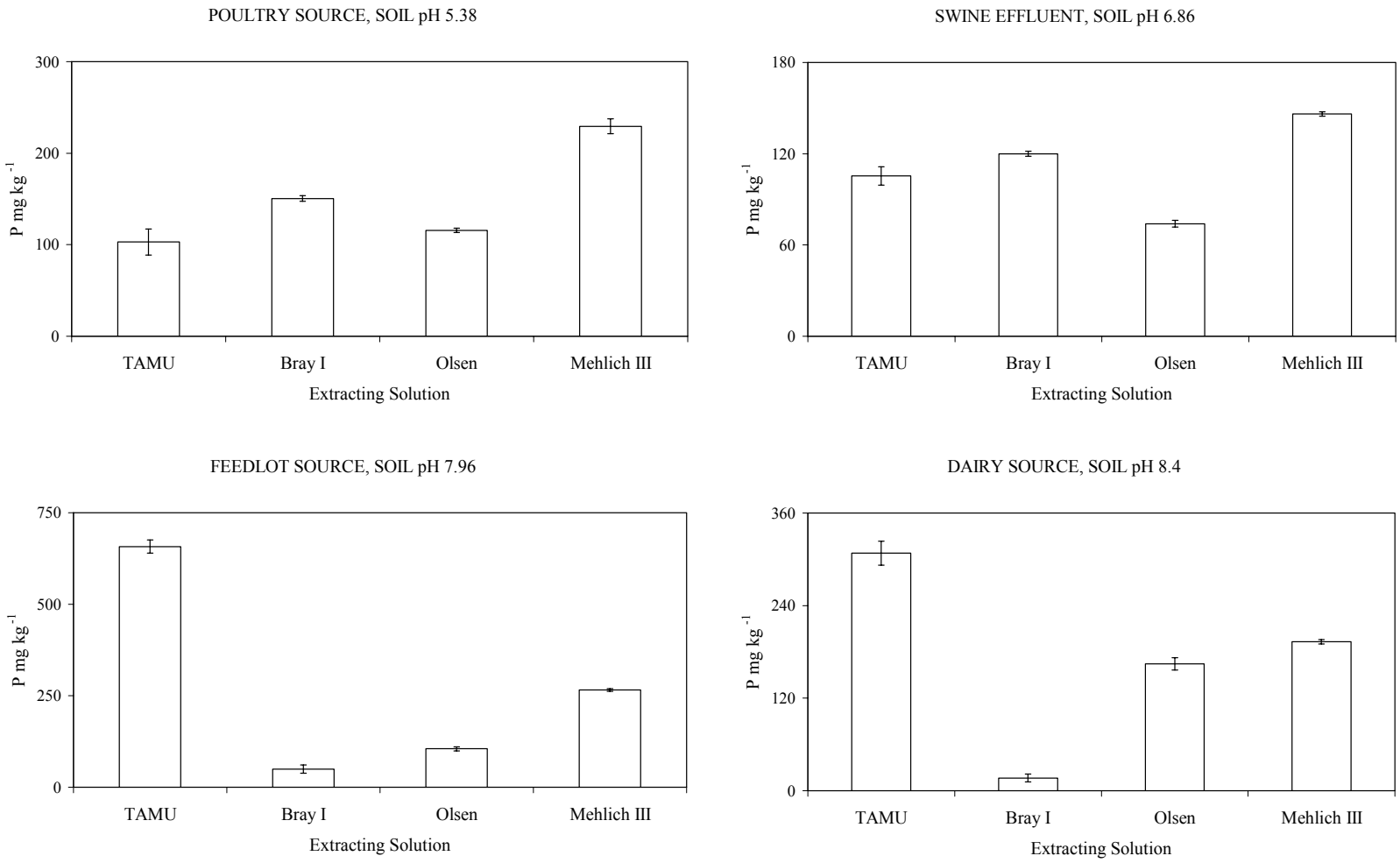

Fig. 2. Effects of manure source on extraction of P by TAMU, Bray I, Olsen, and Mehlich III.

greater concentrations of calcium carbonates (Table 2), the order was altered with the TAMU-solution extracting efficiency gradually rising until it surpassed all others including Mehlich III, which became second in order followed closely by Olsen at the highest soil solution pH. This effect was independent of levels of extractable STP and is illustrated in Figure 3. Meanwhile Bray I extracted P decreased sharply at high soil solution pH. Mehlich III extracted the greatest concentration of P (54.9 – 891.6 mg P kg^{-1} soil) regardless of manure type, soil texture, or rate of application when soil pH was less than 6.9. In acid to neutral soils, Mehlich III STP values could be up to five, three and two times those of the TAMU, Olsen, and Bray I, respectively. When soils had > 130 g kg^{-1} calcium carbonate equivalence (CCE) (Portneuf, Perico and Richfield in Table 2), STP values for Mehlich III increased with respect to Bray I (up to 20 times) and decreased with respect to the TAMU extraction. Reproducibility was consistent throughout all treatments with a CV of 5 or less for most except the Purves soil series (CV 12.2). This increase in variability could be due the high smectitic content present in Purves interfering with the P recovered during the filtration process.

The Bray I extraction (4.4–423.5 mg P kg^{-1} soil) removed the second highest levels of P (usually greater than Olsen but less than Mehlich III in nine of twelve soils when soil pH was less than 6.9. A significant decrease in STP was observed when pH increased to values greater than 7.0, mainly with dairy, swine and feedlot sites. This trend could not be confirmed with the poultry litter, since soil pH did not exceed 6.56 for any of the sites sampled. The extracting solution effects on Bray I were expected due to the neutralization of the dilute HCl in solution by carbonates found in calcareous soils

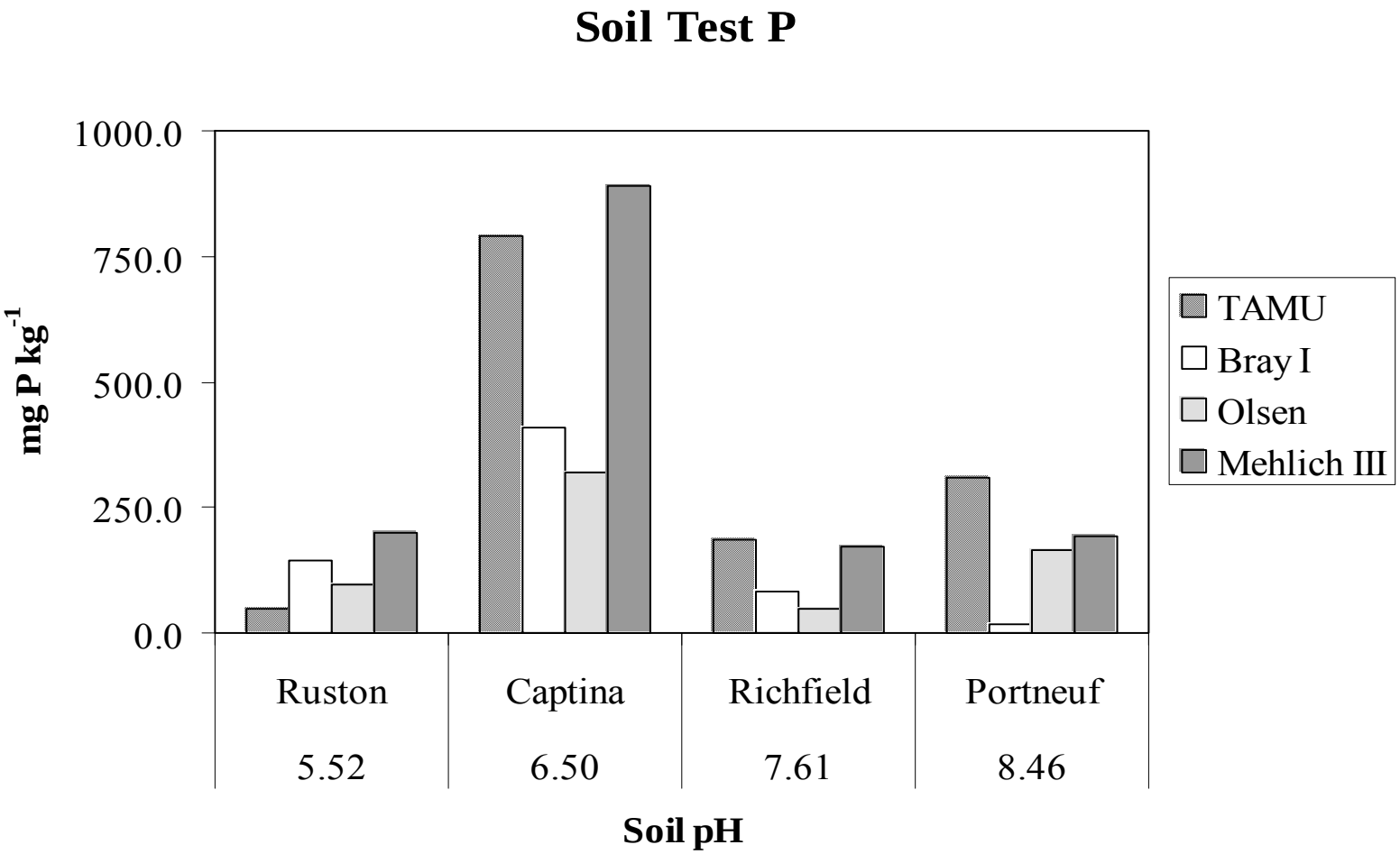

Fig. 3. Effects of soil pH on extracting efficiency of TAMU, Bray I, Olsen, and Mehlich III-P.

(Yee and Broersma, 1987) and the formation of CaF_2 which can precipitate soluble P. In calcareous soils, the low acid concentration in the extract reacts with the calcium carbonate in the soil after which these reactions were not significantly different ($p \geq 0.1$) from the effects of water and the dilute salts. The reproducibility of extractions using the Bray I method decreased sharply when soil pH was above 7.0, with the highest CV (31.5) found in a swine treated site.

Olsen extraction, developed and first used on alkaline and calcareous soils, resulted in significantly lower P concentrations than the Mehlich III P in all but one case, a Purves series which had been treated with dairy manure. However, Olsen STP levels were relatively high (48–429 mg P kg^{-1} soil), even at low soil pH and had little variability. The Olsen extractable P ratio with respect to calcium chloride (Table 5) was 26.2 in the pH range of 5.0- 6.5, increased to 31.6 for 6.0-7.5, and was greatest, 53.8, for values of 7.5 to 8.5. This supports the theory that a layer of P bound to carbonate is being dissolved as the concentration of calcium in solution is reduced by the precipitation of $CaCO_3$. The effectiveness of Olsen on acid soils, which lack carbonates, is not fully understood; although one proposed explanation is that the high OH^- activity decreases Al^{3+} activity by the formation of aluminum hydroxide complexes, and Fe^{3+} by precipitation with oxides, both of which would release the bound P into solution. It is also possible the adsorbed phosphates are being exchanged by the bicarbonate ions. Others have also claimed that a significant portion of the organic bound P fraction is being removed by this extraction (Bowman and Cole, 1978; Tiessen et al., 1983) and, which would be measured by ICP.

Extractions using the TAMU method (NH_4OAc-EDTA) resulted in the lowest extractable P levels of any of the fertility tests (44.3 –887.5 mg P/kg soil) under acid soil conditions, with no observed difference due to soil texture or animal manure source used. As soil pH increased, STP from the TAMU method with respect to Mehlich III increased gradually until they were on the same level. Eventually, under calcareous conditions (pH >7.6), extracting efficiency for TAMU P became as much as three times that of Mehlich III.

Extractable P Ratios Across Soil pH Values

Attempts to create a linear relationship across soil pH for all fertility extractions methods were conducted. All soils regardless of manure source were pooled and their extractable P ratios, with respect to either dilute salt or water extract (Tables 5, 6, and 7), were plotted versus pH ranges 5 – 6, 6 – 7.5, 7.5 – 8.5, and 5 – 8.5. There were no significant linear correlations for any of the methods in any of the pH ranges evaluated. The highest correlation (r^2 = 0.64) occurred with Bray I in the pH range 7.5 – 8.5, a range in which this extraction is not reliable. The method having the highest linear correlation for the pH range of 5.5 to 8.5 was the TAMU (r^2 = 0.39).

Table 6. Extractable phosphorus ratios with respect to potassium chloride, standard deviation (SD), and coefficient of determination (CV) for the pH range.

pH Range		$CaCl_2$†	KCl‡	Water §	TAMU¶	Bray I#	Olsen††	Mehlich III‡‡
5.0 - 8.5	Mean	0.70	1	2.50	35.40	20.90	24.70	42.50
	SD	0.30		1.30	32.70	19.10	23.20	29.80
	CV	45		51	92	91	94	70
	r^2	0.51		0.26	0.15	0.43	0.01	0.14
5.0 - 6.0	Mean	1.00	1	3.50	16.00	35.10	29.40	54.10
	SD	0.20		1.50	6.10	27.00	22.70	40.70
	CV	23		43	38	77	77	75
	r^2	<0.01		<0.01	0.02	0.03	0.02	0.03
6.0 - 7.5	Mean	0.70	1	2.10	27.70	22.70	18.60	39.90
	SD	0.20		1.00	13.80	7.80	8.10	17.80
	CV	34		48	50	35	43	45
	r^2	0.01		0.11	0.34	0.48	0.68	0.56
7.5 - 8.5	Mean	0.50	1	2.00	56.50	8.60	26.50	36.10
	SD	0.20		0.90	44.40	10.80	32.40	30.30
	CV	49		45	79	125	122	84
	r^2	0.47		0.54	0.39	0.64	0.13	0.56

† $CaCl_2$, calcium chloride solution extracted P.
‡ KCl, potassium chloride solution extracted P.
§ Water, deionized water extracted P.
¶ TAMU, ammonium acetate-EDTA extracted P.
# Bray I extracted P
†† Olsen extracted P
‡‡ Mehlich III extracted P

Table 7. Extractable phosphorus ratios with respect to deionized water, standard deviation (SD), and coefficient of determination (CV) for the pH range.

pH Range		$CaCl_2$†	KCl‡	Water §	TAMU¶	Bray I#	Olsen††	Mehlich III‡‡
5.0 - 8.5	Mean	0.30	0.50	1	17.30	8.40	10.50	18.00
	SD	0.10	0.20		16.00	7.20	8.50	12.60
	CV	38	48		93	86	81	70
	r^2	0.11	0.15		0.21	0.22	0.01	0.02
5.0 - 6.0	Mean	0.30	0.30	1	4.70	8.70	7.30	13.50
	SD	0.10	0.20		0.90	4.10	3.00	5.50
	CV	32	56		18	47	41	41
	r^2	0.03	0.01		0.02	0.17	0.06	0.12
6.0 - 7.5	Mean	0.40	0.60	1	17.00	13.60	11.60	24.30
	SD	0.10	0.30		13.40	9.00	10.30	18.50
	CV	40	48		79	66	88	76
	r^2	0.01	<0.01		0.15	0.13	0.18	0.20
7.5 - 8.5	Mean	0.20	0.60	1	26.90	3.50	12.00	15.80
	SD	0.10	0.20		18.40	3.10	9.80	8.70
	CV	23	37		68	88	82	55
	r^2	0.03	0.40		0.12	0.63	0.03	0.39

† $CaCl_2$, calcium chloride solution extracted P.
‡ KCl, potassium chloride solution extracted P.
§ Water, deionized water extracted P.
¶ TAMU, ammonium acetate-EDTA extracted P.
# Bray I extracted P
†† Olsen extracted P
‡‡ Mehlich III extracted P

Conclusions

The source of animal manure and inclusion of organic and other soluble forms of P did not affect extraction efficiency, order, or the reproducibility of extractable P for any of the seven methods used with ICP analyses. Salt extractions removed the least amount of P, while the Mehlich III extracted the most from acid soils, and TAMU extracted the most P from calcareous soils. The difference in fertility tests was likely due to different dissolution rates of the natural apatite and other Ca-bound forms of P in some soils extracted by the TAMU method and Al- and Fe-bound P forms extracted by the Mehlich III method. Variability was low for all procedures except the Bray I extraction when soils were calcareous. Under calcareous conditions, Bray I extracted the same concentrations of P as deionized water but, unlike water extracted P, it produced high variability among replicates. Olsen P, as expected, also proved to be affected by pH, with extraction efficiency with respect to the salt and water extraction methods increasing with soil pH. However, even at low soil pH values, extracted concentrations were relatively high and highly reproducible. Both Mehlich III and the TAMU extractions had very little variance, with typical CV values less than three and four, respectively. Mehlich III consistently extracted the most P compared to all the other tested methods when soil pH was acidic. Under neutral conditions, Mehlich III extracted the same amounts of P as did the TAMU method. The amounts of extractable P using the TAMU method increased linearly with soil pH. At soil pH values above 7.6, TAMU P concentrations were as much as three times those of Mehlich III.

The three methods chosen for rainfall studies: Mehlich III, TAMU and $CaCl_2$ had P extracting efficiencies ranging from low to high. Aside from being highly reproducible, $CaCl_2$ extractions proved easy to filter. The TAMU extract was the primary fertility test used by the Texas Cooperative Extension (TCE) Soil, Water and Forage Testing Laboratory, which was eventually replaced by Mehlich III in January 2004. Comparisons to STP values determined by colorimetric techniques were made in Chapter IV for soils in the High Plains and Central regions of Texas.

CHAPTER III

EFFECTS OF DAIRY MANURE APPLICATION RATES ON SOIL TEST P AND DISSOLVED P IN RUNOFF FROM FOUR BENCHMARK SOIL SERIES IN CENTRAL TEXAS

Synopsis

Some researchers suggest that the quality status of a water reservoir should be dependent upon the total P present. However, significant amounts of particulate P, such as some of the organic bound forms, are so stable that they are unlikely to contribute any P to the system. On the other hand, dissolved P (DP) is the most available form to aquatic plants and can have an immediate impact on water quality. Currently, efforts have been directed to estimate levels of DP coming from fertilized sites using available STP values. Rainfall studies on packed soil boxes have provided a great deal of information on the DP-STP relationship, but are unlikely to represent actual conditions found in open grasslands and pastures. In the case of field experiments, many of these have been conducted primarily on low pH soils, whereas in Texas many of feedyard and dairy CAFOs are located in high pH, calcareous soil regions. Additionally, at the time of this research, only Texas used acidic ammonium acetate (TAMU) as the primary test to determine soil extractable P, which had not been tested for its correlation to runoff P. The North Bosque River area in Central Texas was chosen because of the high concentration of dairies in a watershed that discharges into Lake Waco. Four benchmark soil series with relatively high soil pH were used because it was expected that significant portions of added P would be bound to Ca rather than Al and Fe, which might affect its

solubility in the runoff. Some of these data using colorimetric techniques have been published by Torbert et al. (2002).

Materials and Methods

Site Treatment Prior to Rainfall Simulation

Four soil series representative of major common soil types found in the Bosque and Leon River watersheds in Texas were selected. These soil series were Purves clay, Windthorst sandy loam, Blanket clay loam, and Houston Black clay (Table 1). Six permanent plots were constructed (2 x 3 m) for each soil series with aluminum borders (5 cm above, 10 cm below surface) to collect runoff at each site. All plots had slopes that were approximately 5%. Dairy manure was surface applied to plots at rates of 60, 120, 180, 240, and 360 mg P per kg of soil, in addition to a control that received no application. Following application, manure was allowed to equilibrate with soils for eighteen months prior to rainfall simulations.

Rainfall Application and Collection of Runoff

Constant runoff duration was used instead of constant rainfall duration because results have shown that this provides comparable plot-to-plot runoff amounts and representation of water quality changes throughout the runoff hydrograph (Edward and Daniels, 1993). Vegetation height was maintained between 0.1 and 0.2 m by mowing. Rainfall was applied over plots at an intensity rate of 5 cm per hour using an aluminum frame rain simulator (Tlaloc 3000), with a single low-pressure square pattern nozzle elevated 3.05 m above soil surface. Both height of nozzle and water pressure had been

determined to generate drop-size distribution and terminal velocity (Miller, 1987; Shelton et al., 1985). In order to reduce variability due to moisture conditions, all plots were irrigated prior to simulations and allowed to drain overnight. The simulator was fitted with tarps to protect against wind currents. Three simulated events were performed on each plot, for all rates and soil types. Runoff was collected manually at 5-min intervals for the duration of the runoff trial (30 min). All six discrete runoff samples were pooled into a composite sample, filtered through a 0.45-μm pore membrane, acidified to a pH of about 2 with HCl and frozen until they were analyzed for DP, using a molybdenum-blue method (Murphy and Riley, 1962). Rainfall application was performed by University of Arkansas faculty and staff using their prototype simulator, which was later adopted for use by members of the Southern Extension-Research Activity group for validation of PI in other states. All other simulations in our studies were done with the Texas A&M simulator patterned after the Arkansas prototype.

Soil Sampling and Analysis

Representative soil samples from multiple cores (15 cores, 2.5-cm diameter) were taken from each plot at three different depths (0-2.5 cm, 0-5 cm, and 0-15 cm). The first two were collected after each simulation and sampling holes were filled with clean sand. The 0-15 cm cores were taken after the final rainfall application. Soil extractable P was determined using a number of different extractants. For this study, only soil test P using Mehlich III, TAMU, and calcium chloride referred to as calcium solution soluble P (CSSP) from hereon, which were detailed in Chapter II will be reported. All STP analysis was performed using ICP spectroscopy.

Statistical Analyses

Linear regression analyses of first–order kinetics were used to evaluate the relationships between STP and applied P rates, and runoff DP. First, the Pearson coefficient (r) was used to assess the correlation between applied P and STP for all extraction methods and sampling depths. Coefficient of determination (r^2) between STP and DP in runoff was used to determine the amount of variability that could by explained by linear regression, and to compare best fit due to sampling depth. Analysis of covariance (ANCOVA) was performed to test for equal slopes and x-intercepts between all soils using DP as the dependent variable, STP concentration as the continuous linear effect (covariate), and soil type as the categorical factor that allowed for interaction. Contrast statements on a pair-wise basis were used to test for common slopes for all possible comparisons at each sampling depth using CSSP, Mehlich III, and TAMU-P methods. All analyses were performed using SPSS software Version 11.01 (2001).

Results and Discussion

Soil Extractable P Response to Manure Application Rates

Field incubation of surface applied dairy manure with soil for eighteen months increased extractable P concentrations for the Mehlich III, TAMU, or CSSP methods at every depth sampled (Table 8) for all soil series. Soil test P concentration for Mehlich III and TAMU extractions were different for all soils but close to the applied rates of 0, 60 120, 180, 240 and 360 mg P kg^{-1} at a 0-5 cm depth. By far the greatest levels were

Table 8. Effects of surface manure application on CSSP, Mehlich III, and TAMU soil P extractions after field incubation for 18 months.

Soil Depth (cm)	P added mg / kg	Houston Black				Purves				Blanket				Windthorst			
		CSSP†	Mehlich III	TAMU‡	DP	CSSP†	Mehlich III	TAMU‡	DP	CSSP†	Mehlich III	TAMU‡	DP	CSSP†	Mehlich III	TAMU‡	DP
		— mg / kg —			mg / L	— mg / kg —			mg / L	— mg / kg —			mg / L	— mg / kg —			mg / L
0-2.5	0	1.36	68.4	81.5	0.06	0.2	35.9	41	0.001	5.3	174.1	150	0.242	1.9	40.4	27	0.211
	60	1.85	62.5	151.2	0.15	0.5	95.8	116	0.115	10.7	70.5	49	0.548	4.0	42.7	51	0.401
	120	3.29	164.6	275.0	0.22	1.4	219.4	212	0.281	18.8	-	151	0.558	9.8	154.5	116	0.717
	180	5.01	299.6	436.6	0.30	3.4	482.5	438	0.46	14.6	563.0	463	0.847	9.8	179.6	123	0.801
	240	5.65	393.2	580.3	0.40	5.1	501.7	441	0.412	30.9	982.4	689	1.24	16.8	253.0	194	1.13
	360	10.53	801.6	1065.7	0.41	7.6	677.9	746	0.402	18.4	889.2	598	0.668	23.1	369.4	329	1.45
	r*	0.98	0.96	0.98		0.98	0.97	0.98		0.67	0.90	0.86		0.98	0.98	0.98	
0-5	0	1.78	108.3	75.8	0.06	0.2	13.0	29	0.001	2.6	127.1	113	0.242	0.6	46.9	18	0.211
	60	1.89	46.5	122.3	0.15	0.3	65.6	102	0.115	3.1	49.8	50	0.548	2.9	64.0	35	0.401
	120	2.24	115.7	204.0	0.22	0.6	67.6	98	0.281	9.2	145.0	122	0.558	8.5	157.1	114	0.717
	180	0.30	209.3	311.5	0.30	1.6	316.2	336	0.46	10.8	343.9	306	0.847	8.7	184.2	138	0.801
	240	3.46	218.7	336.3	0.40	3.8	370.7	310	0.412	20.3	418.6	354	1.24	15.4	248.4	194	1.13
	360	3.31	445.0	606.7	0.41	4.1	283.3	430	0.402	10.6	283.0	284	0.668	20.9	409.1	350	1.45
	r*	0.51	0.92	0.98		0.94	0.81	0.94		0.69	0.71	0.78		0.98	0.98	0.98	
0-15																	
	0	1.04	67.8	49.2	0.06	0.2	54.9	28.0	0.001	0.7	47.7	64	0.242	0.2	80.0	12	0.211
	60	1.45	16.3	56.3	0.15	0.2	13.0	50.0	0.115	0.3	23.4	33	0.548	0.2	29.8	12	0.401
	120	1.30	39.2	79.1	0.22	0.2	23.6	61.0	0.281	1.9	41.6	60	0.558	3.5	87.2	80	0.717
	180	1.45	69.8	123.7	0.30	0.2	41.0	77.0	0.46	0.2	116.7	149	0.847	3.4	102.6	91	0.801
	240	1.87	89.6	163.9	0.40	0.4	71.6	114.0	0.412	8.9	157.0	145	1.24	10.1	150.2	136	1.13
	360	7.20	146.9	216.8	0.41	1.2	97.3	189.0	0.402	3.8	97.1	120	0.668	16.1	270.3	240	1.45
	r*	0.82	0.82	0.98		0.84	0.74	0.97		0.58	0.68	0.71		0.95	0.91	0.98	

* All r values were significant at $p < 0.01$
† CSSP, calcium solution soluble P extraction.
‡ TAMU, ammonium acetate-EDTA P extraction.

found on the 0-2.5 cm soil cores. Even though no attempt was made to incorporate the manure, STP values for Mehlich III and TAMU were higher than 200 mg kg^{-1} in the 0-15 cm depth, indicating a significant movement through leaching. While previous results have indicated that the TAMU extracting solution can remove more P than Mehlich III under high soil pH conditions, probably by dissolving apatite, only in the case of the Houston Black soil, which is highly calcareous, was the TAMU solution consistently able to extract more STP at all depths. In the case of the Purves series, which is also highly calcareous, the difference was not as great at the surface (0-2.5 cm) but was still present at deeper depths. Soil test P values of Mehlich III and TAMU for the Windthorst and Blanket series did not show the same degree of difference probably due to the lack of calcareous apatite present. The relationship between STP and added P proved to be linear and highly correlated for each of the four soil series regardless of method of extraction or depth of soil sampled. This showed that, at least in the range of manure applied P (0-360 mg P kg^{-1}), the amount of extractable soil P was dependent upon and proportional to added manure. Data in Figure 4 show the linear relationship for all soils to a depth of 0-2.5 cm with Pearson coefficients ranging from 0.67 to 0.98. Slopes of linear regressions (Table 9) for the top 2.5 cm for Mehlich III ranged from 0.96 to 2.07, TAMU 0.83 to 2.73 and CSSP 0.02 to 0.06. The TAMU solution had the greatest P extracting efficiency as reflected by slope at every depth but only for the Houston Black and Purves series, both of which were highly calcareous, 200 and 400 g kg^{-1}, respectively. In the Blanket and Windthorst soils, Mehlich III consistently extracted more P. The difference may be the result of the TAMU extraction dissolving some calcium bound precipitates of P that are probably more abundant under calcareous conditions.

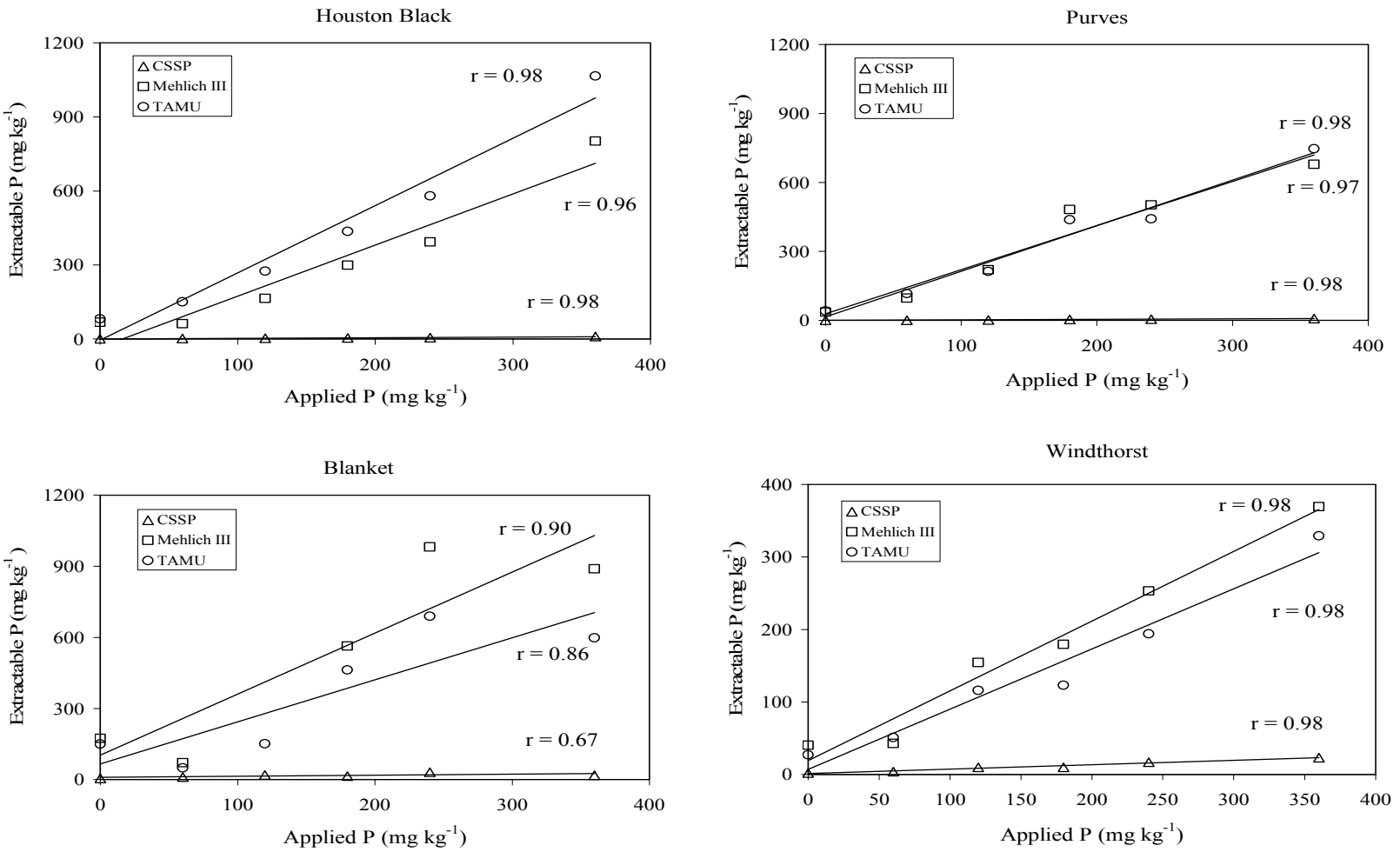

Fig. 4. Linear relationship with Pearson coefficient (r) between applied P and soil test P for three methods to a depth of 2.5 cm for four soil series.

Table 9. Linear regression parameters for the relationship of soil test P and applied dairy manure.

Soil Depth	Soil Series	CSSP†	Mehlich III	TAMU‡
0-2.5 cm	Houston Black	y = 0.0253x + 0.57	y = 2.0713x - 33.093	y = 2.7278x - 4.7414
	Purves	y = 0.0221x - 0.5231	y = 1.9198x + 28.366	y = 1.9833x + 15
	Blankett	y = 0.0448x + 9.2592	y = 2.5749x + 103.24	y = 1.7743x + 66.114
	Windthorst	y = 0.0602x + 1.2733	y = 0.9618x + 19.37	y = 0.83x + 7.2
0-5 cm	Houston Black	y = 0.0045x + 1.4371	y = 1.0009x + 30.439	y = 1.4518x + 43.814
	Purves	y = 0.0128x - 0.2976	y = 0.9677x + 31.239	y = 1.1757x + 29.386
	Blankett	y = 0.034x + 3.9864	y = 0.7839x + 102.48	y = 0.7533x + 84.3
	Windthorst	y = 0.058x + 0.2142	y = 1.0143x + 22.675	y = 0.9207x - 5.8143
0-15 cm	Houston Black	y = 0.0149x - 0.002	y = 0.2845x + 26.077	y = 0.5033x + 34.322
	Purves	y = 0.0026x - 0.0165	y = 0.1783x + 21.7	y = 0.435x + 16.9
	Blankett	y = 0.015x + 0.2281	y = 0.27x + 37.375	y = 0.2695x + 52.043
	Windthorst	y = 0.0465x - 1.8608	y = 0.5817x + 26.937	y = 0.6474x - 8.4143

† CSSP, calcium solution soluble P extraction.
‡ TAMU, ammonium acetate-EDTA extraction.

STP and DP Relationship

Dissolved P in runoff from treated plots ranged from 0.001 to 1.45 mg L^{-1} and increased as extractable soil P increased. Other groups had previously reported this trend for Mehlich III in low pH soils (Pote et al., 1996; 1999) but not for the TAMU extract or $CaCl_2$ at 100 mM. Although these tested soils have high soil pH levels, only Houston Black and Purves are consistently classified as calcareous. Both, the Houston Black and Purves series had STP values equal or greater than the other series, yet DP concentrations in runoff were much lower ranging from 0.001 to 0.4 mg L^{-1} compared to 0.2 to 1.5 mg L^{-1} obtained by Blanket and Windthorst. Establishing the level of significance as $p \leq 0.1$, then plotting STP against DP proved to be significantly correlated for all three methods, soils, and depths (Fig. 5, 6, and 7). There were four cases where the relationships were not statistically significant, with significance declared at $p \leq 0.1$ (Table 10, 11, and 12). Two of these cases occurred within Houston Black at the 0-5 and 0-15 cm depth using CSSP, and two in the Purves series at 0-15 cm depth using Mehlich III and CSSP. Since three of the four cases were at the same depth (0-15 cm), it is likely that this is due to experimental sampling error rather than to a lack of fit to a linear relationship. Another explanation for the lack of significance could be due to calcium-induced precipitation of the extracted P since both soil series involved had high concentrations of $CaCO_3$ and involved the CSSP extractions (100 mM $CaCl_2$). Overall, the STP and runoff DP relationship was more significant when surface soil STP values were used; this was true for all extracting methods. However, TAMU P had a slightly

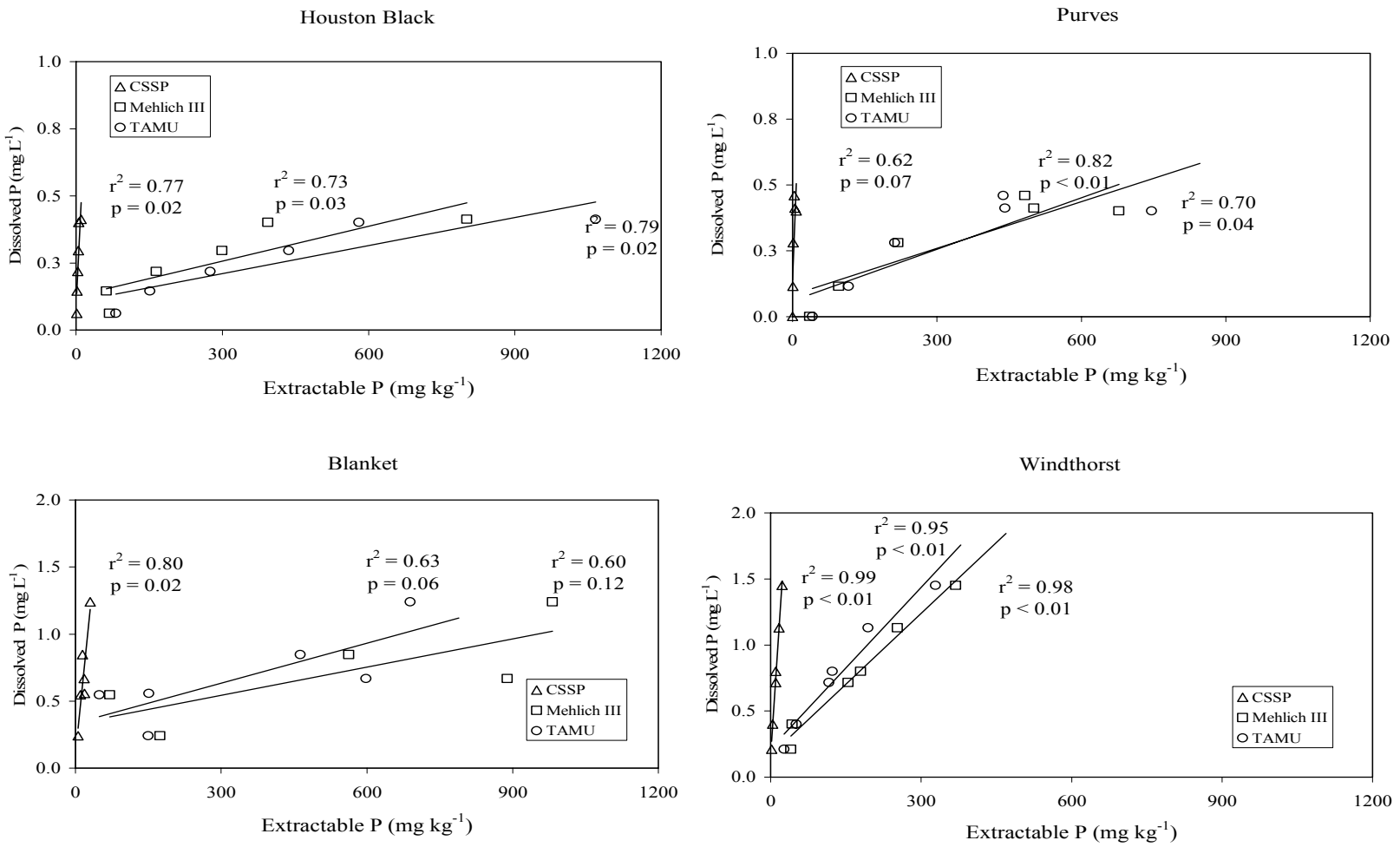

Fig. 5. Coefficient of determination (r^2) between runoff dissolved P and soil test P for three methods to a depth of 2.5 cm for four soil series.

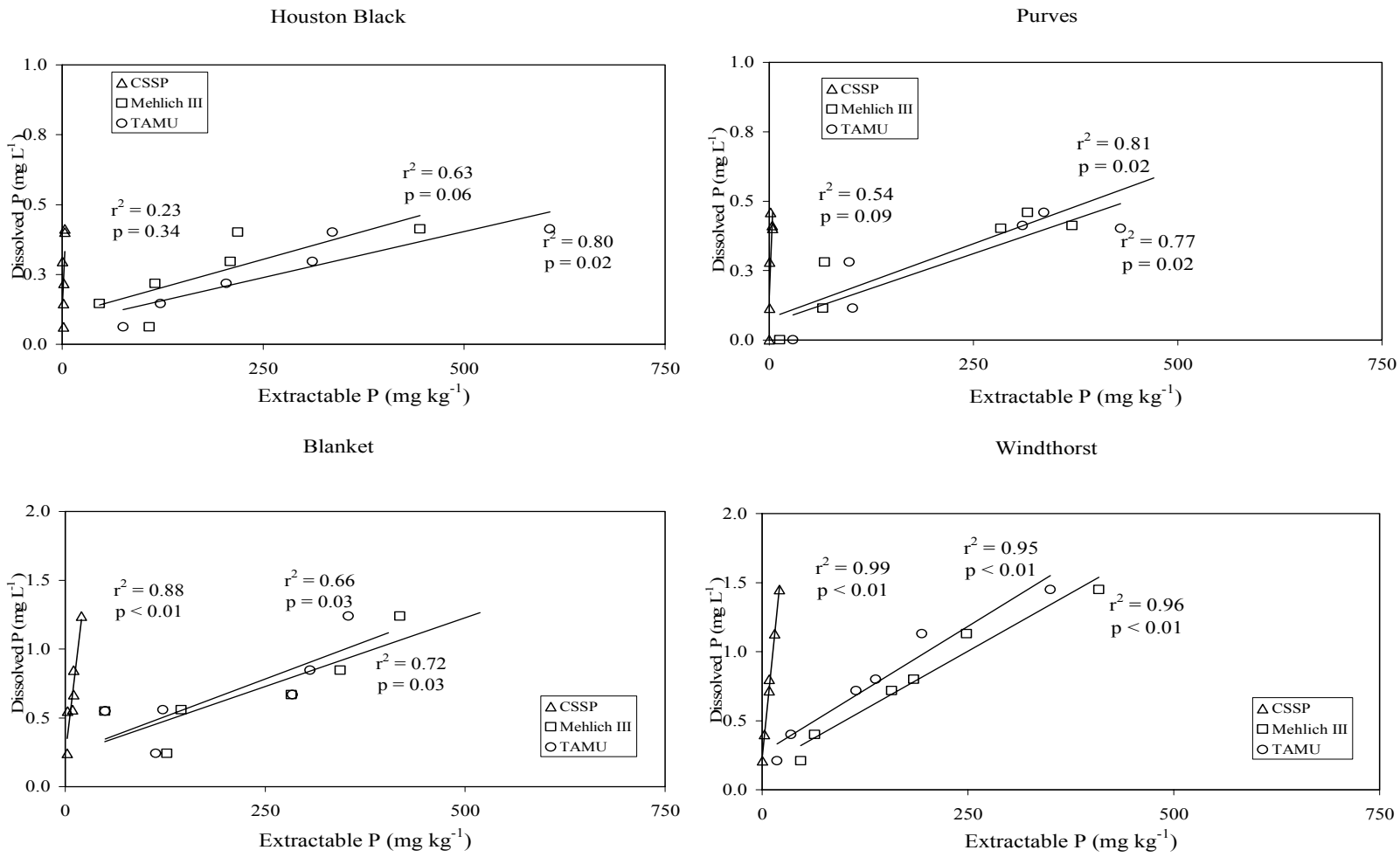

Fig. 6. Coefficient of determination (r^2) between runoff dissolved P and soil test P for three methods to a depth of 5 cm for four soil series.

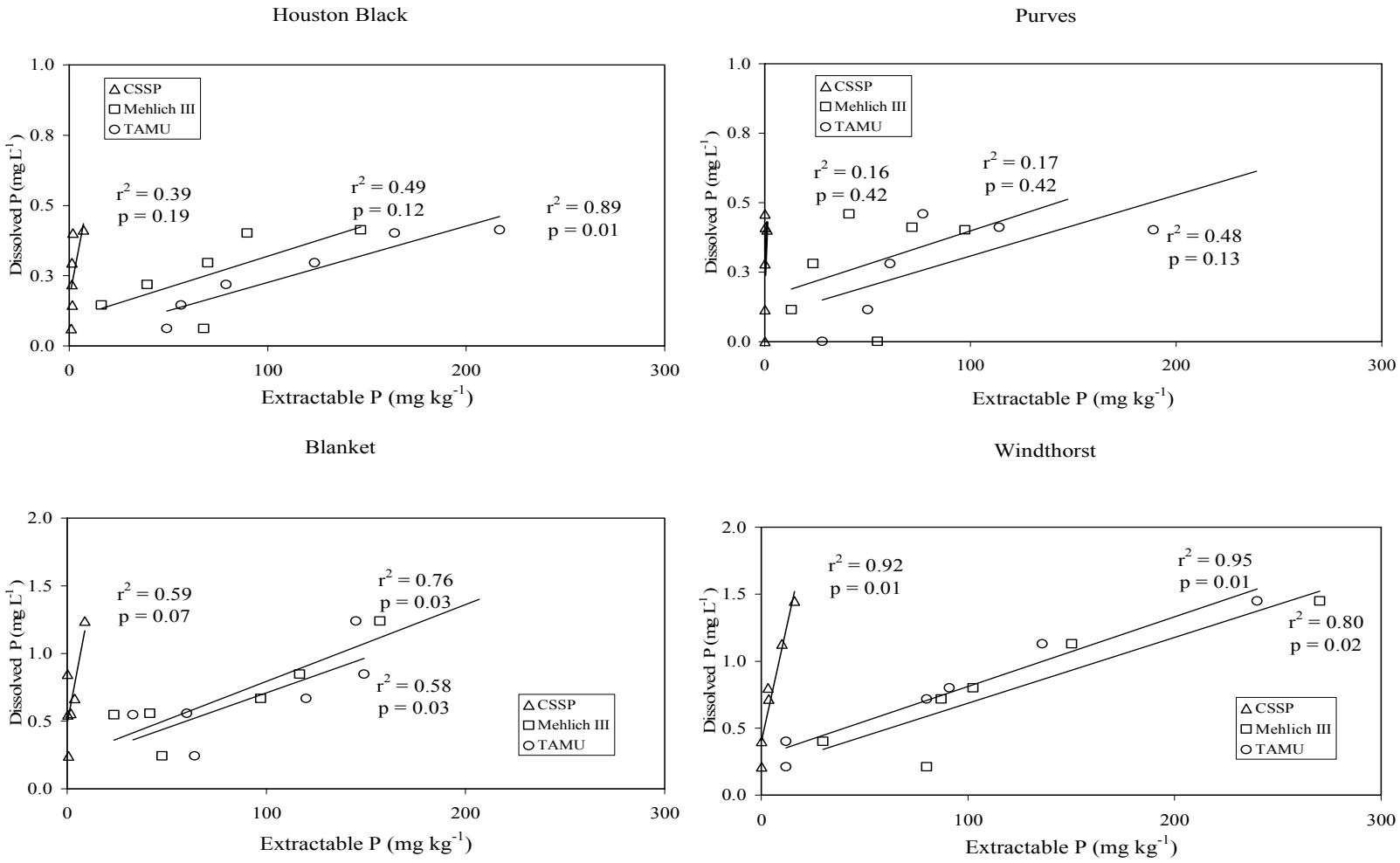

Fig. 7. Coefficient of determination (r^2) between runoff dissolved P and soil test P for three methods to a depth of 15 cm for four soil series.

Table 10. Linear regression parameters and level of significance (p) for the prediction of dissolved P using soil test P for calcium solution soluble P (CSSP) extraction.

Depth (cm)	Soil Series	CSSP	r^2	p	†
0-2.5	Houston Black	$y = 3.66 \times 10^{-2} m + 0.088$	0.77	0.02	a
	Purves	$y = 4.96 \times 10^{-2} m + 0.128$	0.62	0.07	ab
	Blankett	$y = 3.46 \times 10^{-2} m + 0.114$	0.80	0.02	a
	Windthorst	$y = 5.73 \times 10^{-2} m + 0.161$	0.99	< 0.01	b
0-5	Houston Black	$y = 5.72 \times 10^{-2} m + 0.133$	0.23	0.34	a
	Purves	$y = 7.71 \times 10^{-2} m + 0.142$	0.55	0.09	a
	Blankett	y = 4.88 x 10-2 m + 0.224	0.88	< 0.01	a
	Windthorst	$y = 5.99 \times 10^{-2} m + 0.216$	0.99	< 0.01	a
0-15	Houston Black	$y = 3.65 \times 10^{-2} m + 0.170$	0.39	0.19	a
	Purves	$y = 1.88 \times 10^{-1} m + 0.203$	0.16	0.42	a
	Blankett	$y = 7.70 \times 10^{-2} m + 0.481$	0.59	0.07	a
	Windthorst	$y = 6.98 \times 10^{-2} m + 0.396$	0.92	< 0.01	a

	Level of significance for analysis of covariance‡		
	0-2.5 cm	0-5 cm	0-15 cm
STP concentration	< 0.01	< 0.01	0.09
Soil	0.94	0.84	0.14
STP x Soil	0.11	0.68	0.71

† Pair-wise comparison of equations for slopes using significant difference as $p \leq 0.10$.

‡ STP concentration: tests that average slope = 0
Soil: test that intercepts are the same for all soil series
STP x Soil: test that slopes are equal for all soil series

Table 11. Linear regression parameters and level of significance (p) for the prediction of dissolved P using soil test P for Mehlich III extraction.

Depth (cm)	Soil Series	Mehlich III	r^2	p	†
0-2.5	Houston Black	$y = 4.27 \times 10^{-4} m + 0.129$	0.73	0.03	a
	Purves	$y = 6.52 \times 10^{-2} m + 0.060$	0.82	< 0.01	a
	Blankett	$y = 6.99 \times 10^{-4} m + 0.334$	0.60	0.12	a
	Windthorst	$y = 3.58 \times 10^{-3} m + 0.166$	0.98	< 0.01	b
0-5	Houston Black	$y = 7.89 \times 10^{-4} m + 0.106$	0.63	0.06	a
	Purves	$y = 1.07 \times 10^{-3} m + 0.070$	0.81	0.02	ac
	Blankett	$y = 2.00 \times 10^{-3} m + 0.228$	0.72	0.03	c
	Windthorst	$y = 3.36 \times 10^{-3} m + 0.163$	0.96	< 0.01	b
0-15	Houston Black	$y = 2.16 \times 10^{-3} m + 0.102$	0.49	0.12	a
	Purves	$y = 2.40 \times 10^{-3} m + 0.158$	0.17	0.42	ab
	Blankett	$y = 5.67 \times 10^{-3} m + 0.227$	0.76	0.03	b
	Windthorst	$y = 4.91 \times 10^{-3} m + 0.196$	0.80	0.02	ab

	Level of significance for analysis of covariance‡		
	0-2.5 cm	0-5 cm	0-15 cm
STP concentration	< 0.01	< 0.01	< 0.01
Soil	0.35	0.72	0.94
STP x Soil	< 0.01	< 0.01	0.42

† Pair-wise comparison of equations for slopes using significant difference as $p \leq 0.10$.

‡ STP concentration: tests that average slope = 0
Soil: test that intercepts are the same for all soil series
STP x Soil: test that slopes are equal for all soil series

Table 12. Linear regression parameters and level of significance (p) for the prediction of dissolved P using soil test P for ammonium acetate-EDTA (TAMU) extraction.

Depth (cm)	Soil Series	TAMU	r^2	p	†
0-2.5	Houston Black	$y = 3.44 \times 10^{-4} m + 0.108$	0.79	0.02	a
	Purves	$y = 5.90 \times 10^{-4} m + 0.082$	0.70	0.04	ac
	Blankett	$y = 9.94 \times 10^{-4} m + 0.336$	0.63	0.12	c
	Windthorst	$y = 4.07 \times 10^{-3} m + 0.216$	0.95	< 0.01	b
0-5	Houston Black	$y = 6.51 \times 10^{-4} m + 0.077$	0.80	0.02	a
	Purves	$y = 1.00 \times 10^{-3} m + 0.061$	0.77	0.02	ac
	Blankett	$y = 2.18 \times 10^{-3} m + 0.237$	0.66	0.03	c
	Windthorst	$y = 3.67 \times 10^{-3} m + 0.265$	0.95	< 0.01	b
0-15	Houston Black	$y = 1.99 \times 10^{-3} m + 0.028$	0.89	< 0.01	a
	Purves	$y = 2.19 \times 10^{-3} m + 0.089$	0.48	0.13	a
	Blankett	$y = 5.20 \times 10^{-3} m + 0.189$	0.58	0.03	ab
	Windthorst	$y = 5.20 \times 10^{-3} m + 0.290$	0.95	< 0.01	b

	Level of significance for analysis of covariance‡		
	0-2.5 cm	0-5 cm	0-15 cm
STP concentration	< 0.01	< 0.01	< 0.01
Soil	0.31	0.37	0.42
STP x Soil	< 0.01	< 0.01	0.07

† Pair-wise comparison of equations for slopes using significant difference as $p \leq 0.10$.

‡ STP concentration: tests that average slope = 0
Soil: test that intercepts are the same for all soil series
STP x Soil: test that slopes are equal for all soil series

higher level of significance (p=0.04) than Mehlich III (p=0.07) and was much greater than CSSP (p=0.11) across all soils (Appendix A-1).

The level of variability that could be explained by a method as reflected by r^2 varied by method and depth, but was consistent; the greatest difference was due to soil series. For example, all r^2 values in the Purves soil series were lowest regardless of method or depth that was used while those of Windthorst were always highest. The lower correlation of Purves could be the result of greater variability of runoff volume during the simulations, or poor spreading of manure resulting in hot spots during soil sampling. Using averaged r^2 from all soils as an indication of the amount of variability that can be explained, the greatest precision was obtained from soil cores from the surface 0-2.5 cm ($r^2 = 0.78$) or 0-5 cm ($r^2 = 0.75$) than 0-15 cm ($r^2 = 0.60$). This agrees with the concept that during a storm only a thin portion of the surface soil interacts with runoff. Another explanation for the higher variability with depth could be that since the manure was surface applied, only a small portion of the collected soil in the 0-15 core was actually in contact with the added P. Previous reports have indicated that sites with higher STP values might contribute DP at a different rate than those with lower values (McDowell and Sharpley, 2001). However, results for our application rates showed that a first-order kinetic equation described the relationship extremely well in all four soils, and that the increase of DP concentration was proportional to the increase in STP.

Runoff dissolved P concentrations corresponded with increasing STP values as the result of added manure and were represented by slopes of equations. While not equal for all soil series, the effects of soil depth were similar in all soils. Increased depth

of soil cores used for STP determination gave greater slopes in every case. This can be explained as a reduction of the amounts of surface added P as depth increased which in turn yielded lower STP values for all methods. The increase in slope of the regression due to depth can be drastic when CSSP values are being used. Using the Purves series to illustrate this point, at a depth of 0-2.5 cm the calculated slope was 4.96×10^{-2} but it increased to 1.88×10^{-1} when depth was 0-15 cm. This can create problems making reliable estimates, was reflected by observed changes in variability (r^2 was reduced from 0.62 to 0.17), and would require a change in the sampling techniques currently being recommended. This same effect on the Purves series was not as drastic when values from the TAMU solution were used, probably because extracting P concentration were much larger. Although, it is also possible that by sampling down to 15 cm we were diluting the amount of P in the soil that was in direct contact with the applied manure, thereby increase variability. We could not make this interpretation for all soils. The increase in variability by depth of soil sampling which we recorded for surface application might not be a significant concern with dairies located in the Bosque watershed, where current practices for most CAFOs are to haul solids offsite and to apply only effluent.

Regression equations between STP and runoff DP for all soil types at each depth were compared for each extracting method in Tables 10-12. There was a statistically significant response in DP to the increase of STP concentration at every depth for all extraction methods, meaning that slope was not zero for any of the four soil series that were used. This was present regardless of method used to determine STP. Comparisons

of the regression equations were performed for each soil by analyzing the interactions of P concentration and soil type (STP*Soil Series) at each depth. Calcium chloride-extracted P (CSSP) showed that the regression equations were not significantly different at any depth when significance was established as $p \leq 0.1$. However, the level of significance for different slopes was only 0.11 at the 0-2.5 cm depth (Table 10). In addition, pair-wise comparisons at the same 0-2.5 cm depth also showed that Windthorst was significantly different from all other soil series except Purves. No difference in slopes among soil series for any other depth could be detected. The lack of difference in slopes or intercepts for the regression lines between soil series at the 0-5 and 0-15 cm depths could be interpreted as evidence that a single relation can be created to estimate DP in runoff for all soils if deep sampling is used, at least for the calcium chloride-extractable P. However, given the observed differences between soil types at the 0-2.5 cm depth, it is also possible that the lack of differences is actually an inability to describe the STP to DP relationship at deeper soil sampling depths.

Mehlich III STP (Table 11) was better able to differentiate among soil equations than CSSP. Significant differences in the slopes of linear equations could be detected at the 0 to 2.5 and 0 to 5-cm depth but not at 0 to 15-cm. Pair-wise comparison showed that the greatest sensitivity occurred at the 0 to 5-cm depth, where differences among three soils could be made. One explanation for the lack of difference at the 0 to 2.5-cm depth could be due to hotspots and the difficulty of accurately and consistently sampling such a shallow core. At the 0 to 15-cm depth, the difference might be due to variations introduced by dilution of the amount of P that is in contact with the runoff.

Comparisons using TAMU STP (Table 12) proved to be the most sensitive based on the levels of significance. Clear differences were obtained among regression equations (STP*Soil Series) for samples at all depths. Comparison among soils also showed a difference among three soil series; however sensitivity decreased with depth where at the 0-15 cm only one soil series was clearly different.

One conclusion from these experiments is that STP values for Mehlich III and TAMU extractions in the application range (0-360 mg P kg^{-1}) are less likely to be affected by variability introduced from sampling hot spots in plots. On the other hand, CSSP are more susceptible to minor extractable changes by soil test methods because of the low P values, which only ranged from 0.2 to 23 mg kg^{-1}. The use of CSSP does appear to provide a good basis for making multiple soils predictions of the amount of DP expected to be released.

A final effort to obtain a single linear relationship for DP and STP was attempted by pooling all soils together. The result, as expected, was a drastically decreased relationship for all Mehlich III and TAMU extractions, but not for CSSP as reflected by r^2 (Fig. 8, 9 and 10). This negative effect is not surprising since there is such a drastic difference in the levels of DP released by soils at the same STP value. For example, the

surface soil of Houston Black clay with a Mehlich-III P value of 300 mg kg was found to release 0.3 mg L^{-1} DP while Windthorst at the same STP released 1.4 mg L^{-1}. With both TAMU and Mehlich III extractants, the level of correlation was lowest for surface soil and increased with depth. The greater level of correlation for the deeper soil samples was expected since comparison of linear equations at the 0-15 cm depth showed no statistical differences between soils. Thus, the use of a single relationship to describe release of surface soil P into runoff, as DP, cannot be used for the four soil series chosen. These results agreed with those of Turner et al. (2004), who were also unable to obtain a single relationship between 3 calcareous soil series extracted with 0.01 mm $CaCl_2$. They proposed that the differences were due to $CaCO_3$ content, 22-245 mg kg^{-1}, extractable Ca, 100-490 mg kg^{-1}, and Fe, 0.01 – 15.9 mg kg^{-1}. Whether this relationship holds for soil under more acidic soil pH conditions will be examined in the following chapter.

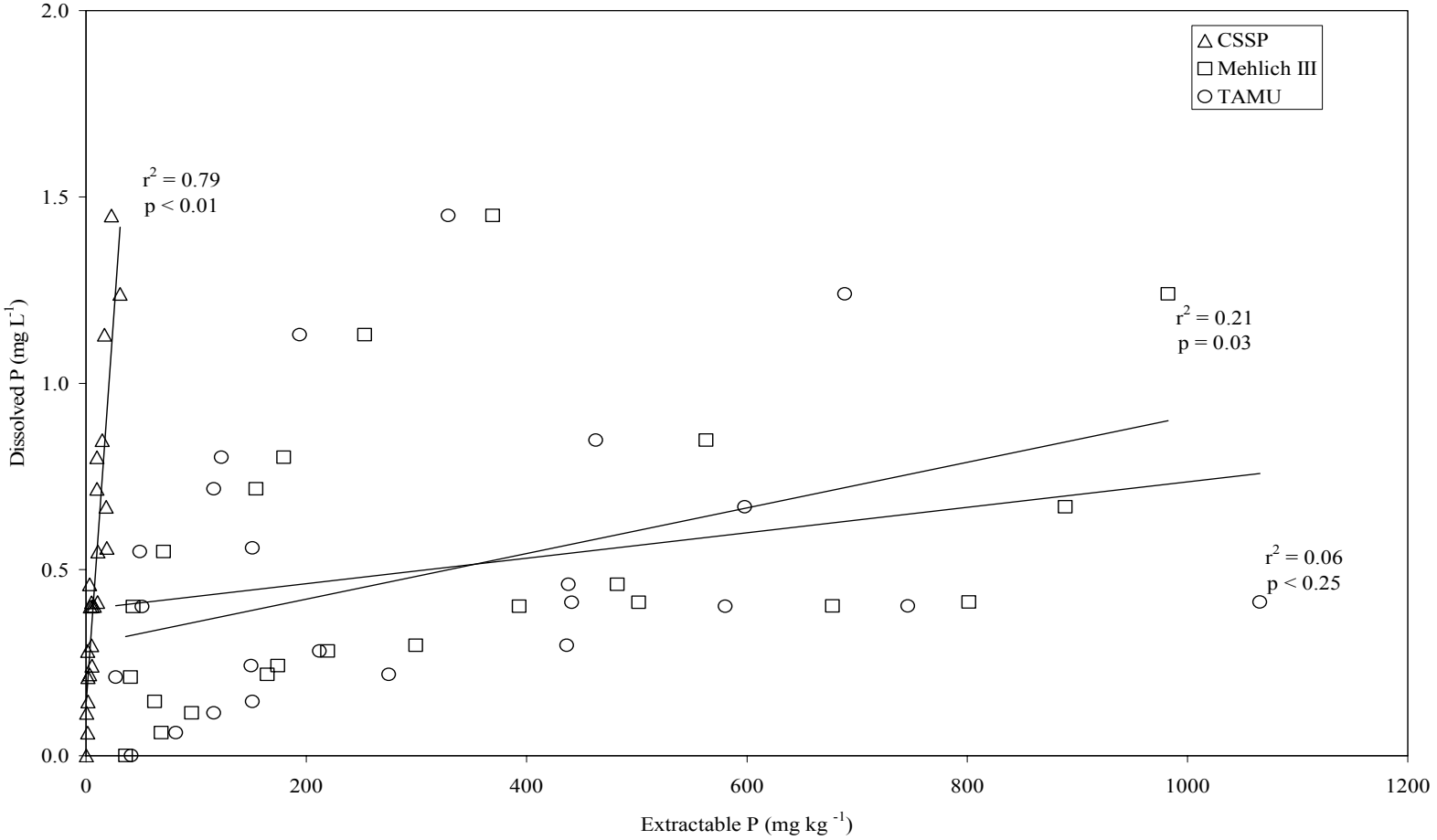

Fig. 8. Relationship between extractable soil test P (0-2.5 cm) for calcium solution soluble P (CSSP), Mehlich III, and ammonium acetate-EDTA (TAMU) with dissolved P in runoff for all soil series grouped together.

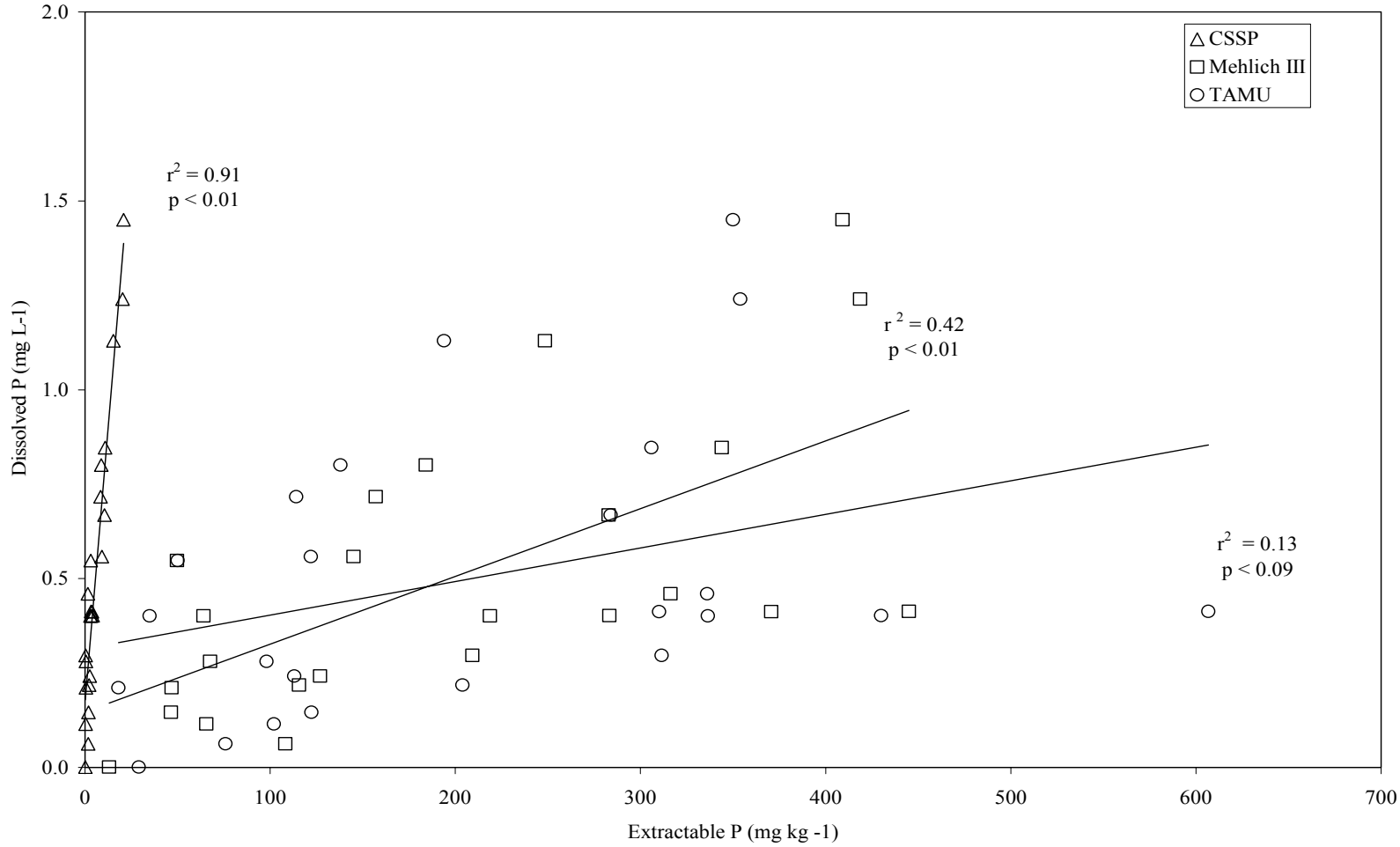

Fig. 9. Relationship between extractable soil test P (0-5 cm) for calcium solution soluble P (CSSP), Mehlich III, and ammonium acetate-EDTA (TAMU) with dissolved P in runoff for all soil series grouped together.

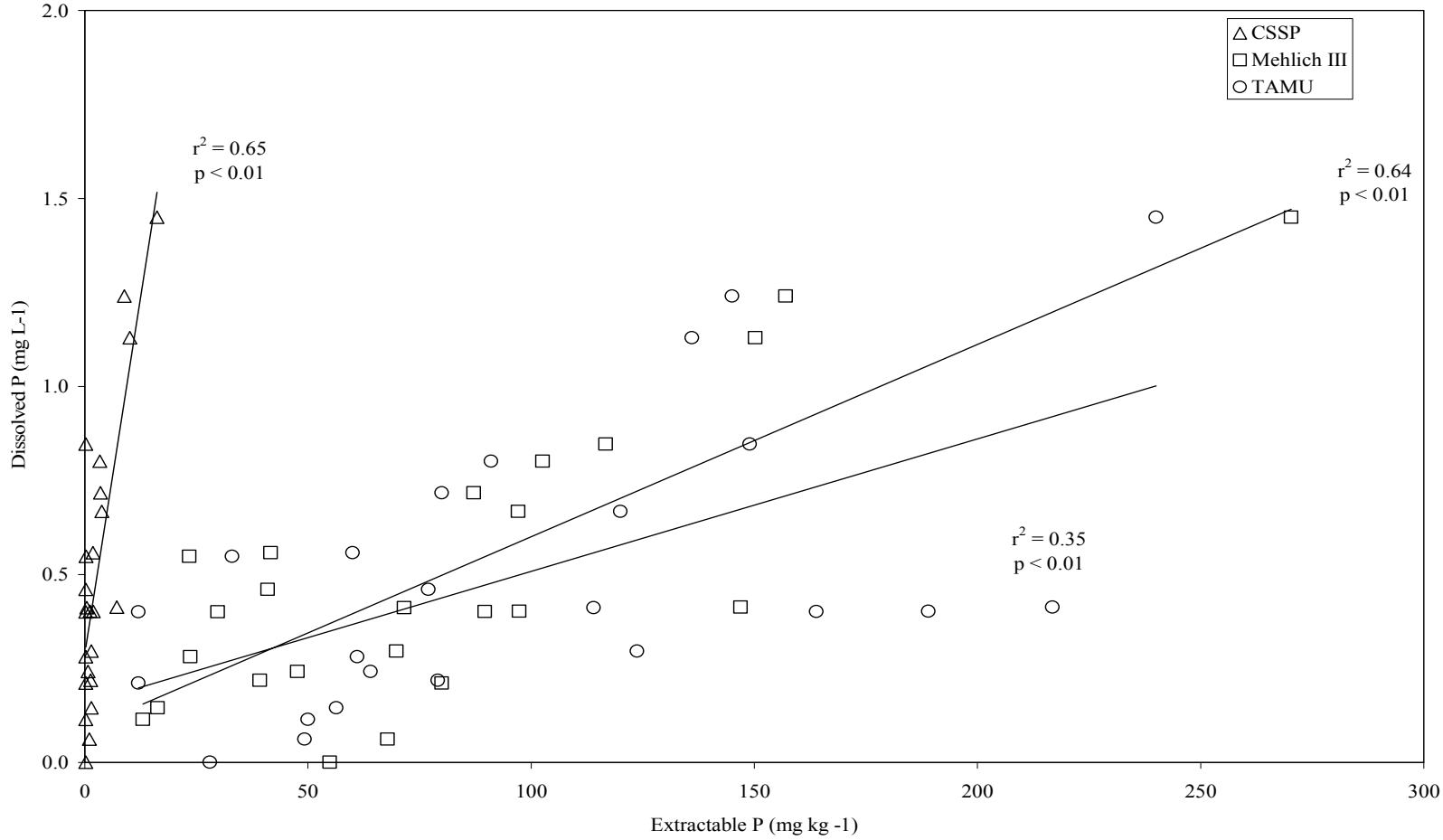

Fig. 10. Relationship between extractable soil test P (0-15 cm) for calcium solution soluble P (CSSP), Mehlich III, and ammonium acetate-EDTA (TAMU) with dissolved P in runoff for all soil series grouped together.

Conclusions

This experiment showed that extractable P for the TAMU, Mehlich III, and CSSP soil testing methods were highly correlated to the application rates of dairy manure in four major soil series from the Bosque region of Central Texas. Rainfall simulations proved that first order kinetics could be used to predict concentrations of dissolved P in runoff in plots through the use of $CaCl_2$, Mehlich III, and TAMU soil extractable P. This relationship was not restricted to STP values obtained by collection of soil from the surface but extended for cores down to a depth of the traditional 15 cm used for most soil nutrient analysis. While the use of greater sampling depth for STP values of CSSP or Mehlich III showed no statistical difference in the relationship to DP, this was probably the result of higher variability rather than a similar behavior among soil series. The use of TAMU P gave significant differences between soil series at all soil depths. Because manure was only surface applied, relationships between runoff DP and STP were better correlated for surface soil samples (0-2.5 cm) and decreased as depth of soil sampling increased. However, correlations obtained for 0-15 cm depths were not much lower and still high so that current requirements to sample 0-5 cm might not be necessary, especially since a large number of CAFOs in Texas, either incorporate manure to depths greater than six-inches, or only apply effluent on site, which results in P being more evenly distributed though out the soil profile. These results also showed that dissolved P released to runoff could be very different for soils even when they had similar amounts of manure applied and had similar STP values. Thus, attempts to

implement a universal environmental P soil test would seem unwise, at least when soil solution pH is high.

CHAPTER IV

DETERMINATION OF P LOSSES FROM CAFO FIELDS

Synopsis

Previous experiments in chapter III showed that STPs for four benchmark soils in Central Texas were correlated to surface application rates of dairy manure. Furthermore, the use of ICP rather than colorimetric techniques for soil P analysis did not affect the linear first order kinetic relationship of using STP to predict DP in the runoff. However, this relationship proved to be soil series specific. For these trials we selected 23 sites from four counties, extending from the Southern High Plains to Central Texas. These sites were all in working CAFOs with different management practices. No attempt was made to maintain a constant period between time of manure or effluent application and rainfall simulation. The goal was to determine if P losses in the runoff could be related to STP values under working CAFO conditions. All P analysis involved ICP techniques with three soil-extracting methods.

Materials and Methods

Site Selection

Sites within working feedlots CAFOs in the Texas High Plains and dairies in the Leon and Bosque watersheds were surveyed to encompass a number of properties including slope, management practices, distance to nearest named stream, soil pH, and soil extractable P. All but two sites have had manure or effluent applied at some time. Composite soil sample cores were taken at a depth of 0-15 cm from each site and

analyzed for STP using the TAMU and Mehlich III method. Five sites in the Southern High Plains and eighteen in the Central Texas regions were selected.

Treatment of Water Source

In order to maintain water quality, all water used except for prewetting purpose was filtered through two mixed bed columns of cation and anion resins. A final cation column was used to ensure that water pH was maintained between 4.8 and 6, so that conditions would be similar to natural rainfall. The conductivity and pH of our water was periodically monitored in the field before and after every simulation. Whenever conductivity of water approached 20 μS, new columns were used. Water samples were collected daily and analyzed for NO_3^--N, B, Na, Mg, P, S, K, Ca, Mn, Fe, Cu, and Zn content using ICP techniques (Provin, 2003).

Rainfall Application and Collection of Runoff

Plot locations were recorded using Global Positioning System (GPS). Three nonpermanent metal frames (1.5m x 2 m) were driven into the ground with the long axis facing down slope to provide three replicates. Vegetation when present was mowed to a height of 20 cm and clippings removed prior to simulation. Figure 11 provides a view of the simulator frame over the plots. In order to reduce variability due to moisture conditions, all plots were irrigated prior to simulations by low-intensity sprinkler at a rate of about 2.69 cm h^{-1} until the surface appeared saturated, and then allowed to drain overnight. Because of the frame design, any water that accumulated was constrained within the metal frame. The following day volumetric water contents (θw) were taken

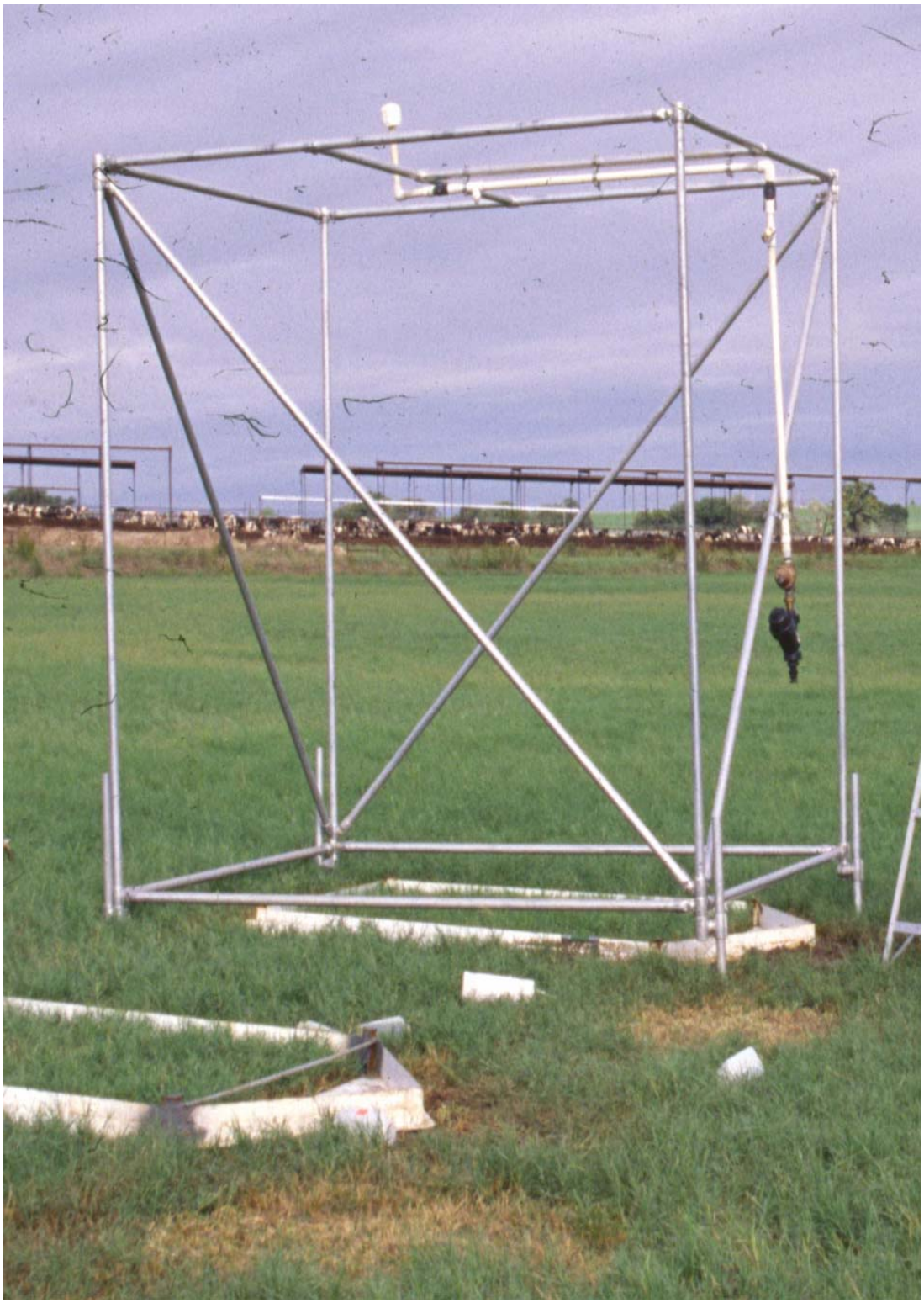

Fig. 11. Tlaloc 3000 rainfall simulator and frames used to collect runoff.

using a soil moisture probe (ThetaProbe ML2) to ensure that soils were close to field capacity (based on soil texture), before rainfall was applied to each specific soil (Daniel, 1999). Rainfall was applied over plots at an intensity rate of 7.5 cm per hour (intensity of a 1-hr/10-yr storm event in Stephenville, TX, used on the P Benchmark Soil Project) using an aluminum frame rain simulator (Tlaloc 3000), with a single low-pressure square pattern nozzle elevated 3.05 m above soil surface (Fig. 11). Both height of nozzle and water pressure had been determined to generate drop-size distribution and terminal velocity (Miller, 1987; Shelton et. al., 1985). Rainfall duration was for 30 minutes following the generation of continuous runoff. The simulator was fitted with tarps to protect against wind currents. Three simulated events were performed on each plot. Time between simulations on an individual plot ranged from 2 to 24 hr. Runoff volume collected in the frame's flume was transferred into plastic containers using electrical pumps. Total runoff and flow was recorded using digital balances and then transferred into a 121 L container. Three 1 L samples, one each at fifteen and thirty minutes after initial runoff, and one composite at the end of the run were collected in polyethylene storage bottles. An additional composite sample was collected for NO_3^--N analysis only. Conductivity and solution pH were taken for each sample in the field, then acidified with HCl (except for the nitrate-N samples) and stored in the dark at 4^o C. Fifteen mL were filtered through a 0.45 μm pore membrane and analyzed using ICP within a week from the time they were collected, for dissolved P, Ca, Mg, Na, K, S, B, Mn, Fe, Cu, and Zn. All samples were analyzed for dissolved nutrients using ICP except for NO_3-N, which was reduced to nitrite and then measured colorimetrically (Provin, 2003). Twenty-five

mL of unfiltered solution were digested for total P in runoff determination (Pote and Daniel, 2000). Total soil P was measured by digesting 1 g of soil with 10 mL nitric acid at 125 ^{o} C until sediment became colorless (Feagley et al., 1994).

Soil Sampling and Analysis

Representative soil samples from multiple cores (2.5 cm diameter) were taken from each plot at three different depths (0-5 cm, 5-15 cm, and 0-15 cm) following the third rainfall application. Soil extractable P was determined using the Mehlich III, TAMU, and $CaCl_2$ (CSSP) solution detailed in Chapter II. Analysis of STP was performed with ICP spectroscopy for all depths, and colorimetric techniques were also used on the 0-15 cm depth samples using the Mehlich III and TAMU extractions (Dick and Tabatabai, 1977). Calcium carbonate equivalent (CCE) and inorganic carbon (IC) were determined using a Chittick apparatus (Dremanis, 1962). Soil organic carbon analysis was performed by furnace combustion at 660 C (Provin, 2003).

Statistical Analyses

Soil series were grouped into acidic, neutral, and calcareous soil pH ranges, and then first order linear regression analysis were performed at three soil sampling depths between STP and DP or TP in runoff. Coefficient of determination (r^2) between STP and DP or TP in runoff was used to determine the amount of variability that could by explained by linear regression, and compare differences due to sampling depth.

Results and Discussion

Hydrological Properties

Because of the wide range of soils and management properties, one would expect different hydrological characteristics for many of the sites. Results by other researchers have indicated that first runs, equivalent to the first storm events of the year, give the most variable results from plot to plot, and that more reproducible data can be collected from later runs (Daniel, 1999). In an attempt to address this, we decided to perform multiple simulations on each of our site replicates. Hydrologic parameter means and CV averaged for three replications per site, with three runoff events on each plot are listed in Table 13, while those averaged for only the first run can be found in Table 14. The overall CVs for the amount of rainfall needed to generate continuous runoff, total rainfall applied, runoff volume depth, and total particulate matter eroded with runoff were decreased in most cases when only the first simulation event was used. If an average CV is calculated for all plots we see an overall reduction in the variability of rainfall added, runoff depth, total rainfall applied, and particulate matter removed from 41, 16, 6, and 50 to 29, 18, 5, and 44, respectively. One reason for the variability could be different initial moisture contents due to a longer draining period between the initial wetting of plots and first simulation (24 hr) versus the period for the second and third runs (2 hr to 24 hr). As a result, subsequent runs might have had higher moisture content at the start of the rainfall application. Another physical factor influencing results is the greater removal of small soil colloids and other debris during the first rainfall event, especially when no surface cover was present. For example more clay-sized

Table 13. Hydrological parameters using multiple rainfall simulations on each plot for all simulations.

Plot	Slope	Time to Runoff (sec) Average	SD	CV	Rainfall Applied (mm) Average	SD	CV	Runoff Volume (L) Average	SD	CV	Runoff Depth (mm) Average	SD	CV	Total Rainfall Applied (L) Average	SD	CV	Total Sediment (g) average	SD	CV
Plot A	1.23	228	174	76	4.8	3.6	76	132	11	8	44.0	3.7	8	42.3	3.6	9	206.27	52.60	25
Plot B	5.79	83	35	43	1.7	0.7	43	146	7	5	48.5	2.3	5	39.2	0.7	2	714.40	371.88	52
Plot C	0.18	359	267	74	7.5	5.6	74	77	28	36	25.7	9.2	36	45.0	5.6	12	29.32	13.65	47
Plot D	0.28	265	126	48	5.5	2.6	48	112	23	21	37.4	7.7	21	43.0	2.6	6	253.51	66.45	26
Plot E	0.20	234	172	74	4.9	3.6	74	113	19	17	37.5	6.4	17	42.4	3.6	8	398.79	188.06	47
Plot F	4.54	133	52	39	2.8	1.1	39	131	18	14	43.6	5.9	14	40.3	1.1	3	359.84	114.49	32
Plot G	0.81	198	81	41	4.1	2.1	51	102	36	35	34.0	11.9	35	41.6	2.1	5	92.15	31.75	34
Plot H	4.41	909	842	93	18.9	17.5	93	101	35	34	33.8	11.6	34	56.4	17.5	31	5.96	3.26	55
Plot I	3.89	764	147	19	15.9	3.1	19	92	14	15	30.8	4.6	15	53.4	3.1	6	58.81	25.17	43
Plot J	4.16	137	25	18	2.8	0.5	18	139	5	3	46.4	1.5	3	40.3	0.5	1	142.79	48.65	34
Plot K	3.73	201	119	59	4.2	2.5	59	149	4	3	49.5	1.4	3	41.7	2.5	6	63.04	30.76	49
Plot L	2.15	232	107	46	4.8	2.2	46	112	15	13	37.4	4.9	13	42.3	2.2	5	238.71	109.38	46
Plot M	7.59	354	103	29	7.4	2.2	29	103	14	13	34.4	4.6	13	44.9	2.2	5	113.97	52.93	46
Plot N	6.71	439	57	13	9.2	1.2	13	111	8	8	37.2	4.6	12	46.7	1.2	3	62.93	37.93	60
Plot O	5.85	642	122	19	13.4	2.5	19	59	18	31	19.8	6.1	31	50.9	2.5	5	0.00	0.00	
Plot P	8.06	466	64	14	9.7	1.3	14	77	19	24	25.7	6.3	24	47.2	1.3	3	0.00	0.00	
Plot Q	10.26	95	35	36	2.0	0.7	36	133	5	4	44.4	1.7	4	39.5	0.7	2	717.47	273.32	38
Plot R	4.48	362	78	22	7.5	1.6	22	101	30	30	33.6	10.1	30	45.0	1.6	4	6.04	6.96	115
Plot S	2.11	320	46	14	6.7	1.0	14	142	11	7	47.2	3.5	7	44.2	1.0	2	10.19	6.43	63
Plot T	3.01	620	203	33	12.9	4.2	33	123	11	9	40.8	3.7	9	50.4	4.2	8	4.98	7.01	141
Plot U	1.92	183	45	24	3.8	0.9	24	141	9	7	47.0	3.1	7	41.3	0.9	2	167.26	31.22	19
Plot V	4.25	93	64	69	1.9	1.3	69	134	14	10	44.6	4.5	10	39.4	1.3	3	881.23	374.18	42
Plot W	2.46	553	217	39	11.5	4.5	39	151	11	7	50.3	3.7	7	49.0	4.5	9	400.98	138.65	35
Averaged CV				41			41			15			16			6			50

Table 14. Hydrological parameters using only the first rainfall on each plot.

Plot	Slope	Time to Runoff (sec) Average	SD	CV	Rainfall Applied (mm) Average	SD	CV	Runoff Volume (L) Average	SD	CV	Runoff Depth (mm) Average	SD	CV	Total Rainfall Applied (L) Average	SD	CV	Total Sediment (g) average	SD	CV
Plot A	1.23	343	284	83	7.1	5.9	83	124	2	2	41.4	0.7	2	44.6	5.9	13	230.43	10.59	5
Plot B	5.79	124	23	18	2.6	0.5	18	147	5	4	49.0	1.8	4	40.1	0.5	1	1173.13	194.85	17
Plot C	0.18	711	66	9	14.8	1.4	9	51	29	56	17.1	9.6	56	52.3	1.4	3	20.98	10.95	52
Plot D	0.28	228	145	64	4.8	3.0	64	106	32	30	35.2	10.7	30	42.3	3.0	7	244.14	53.05	22
Plot E	0.20	386	199	52	8.0	4.1	52	93	2	2	31.1	0.6	2	45.5	4.1	9	385.59	203.73	53
Plot F	4.54	199	29	15	4.1	0.6	15	121	26	21	40.2	8.6	21	41.6	0.6	1	477.92	90.88	19
Plot G	0.81	257	85	33	5.4	1.8	33	64	33	52	21.5	11.1	52	42.9	1.8	4	90.17	28.08	31
Plot H	4.41	1777	1064	60	37.0	22.2	60	63	34	54	20.9	11.2	54	74.5	22.2	30	3.01	4.27	142
Plot I	3.89	893	46	5	18.6	1.0	5	89	11	12	29.7	3.6	12	56.1	1.0	2	80.61	15.75	20
Plot J	4.16	143	24	17	3.0	0.5	17	137	5	4	45.6	1.7	4	40.5	0.5	1	197.29	13.12	7
Plot K	3.73	334	94	28	7.0	2.0	28	145	4	3	48.4	1.3	3	44.5	2.0	4	99.75	14.63	15
Plot L	2.15	298	142	48	6.2	3.0	48	106	15	14	35.5	4.8	14	43.7	3.0	7	273.75	130.43	48
Plot M	7.59	403	126	31	8.4	2.6	31	90	10	12	29.9	3.4	12	45.9	2.6	6	103.18	39.13	38
Plot N	6.71	492	47	10	10.3	1.0	10	97	8	9	32.5	2.8	9	47.8	1.0	2	56.73	36.88	65
Plot O	5.85	593	109	18	12.4	2.3	18	58	18	31	19.3	6.1	31	49.9	2.3	5	0.00	0.00	
Plot P	8.06	484	83	17	10.1	1.7	17	77	21	27	25.8	6.8	27	47.6	1.7	4	0.00	0.00	
Plot Q	10.26	126	29	23	2.6	0.6	23	133	5	4	44.2	1.7	4	40.1	0.6	2	778.19	423.96	54
Plot R	4.48	439	19	4	9.2	0.4	4	73	31	43	24.2	10.3	43	46.7	0.4	1	8.36	10.07	120
Plot S	2.11	317	62	19	6.6	1.3	19	134	13	10	44.7	4.5	10	44.1	1.3	3	18.33	3.35	18
Plot T	3.01	692	229	33	14.4	4.8	33	120	7	6	39.9	2.3	6	51.9	4.8	9	5.54	6.46	117
Plot U	1.92	215	23	11	4.5	0.5	11	132	4	3	44.0	1.4	3	42.0	0.5	1	188.16	20.49	11
Plot V	4.25	158	64	41	3.3	1.3	41	117	5	4	38.9	1.7	4	40.8	1.3	3	914.02	503.55	55
Plot W	2.46	715	148	21	14.9	3.1	21	152	6	4	50.6	2.0	4	52.4	3.1	6	556.68	105.37	19
Averaged CV				29			29			18			18			5			44

micelles will be expected in the first run, leaving heavier particles behind that will result in a net decrease of sediment eroded for subsequent events. This also can affect the amount of vegetative residue being removed with the runoff. In most of our trials we saw a decrease in total particulate matter eroded for later runs as reflected by the lower values when all three runs were averaged (Table 13). Averaging all runs for each plot yielded lower values for the amount of rainfall needed to produce runoff, total rainfall applied, and particulate matter removed, while giving higher amounts of runoff depth than the parameters calculated with only the first runoff event.

Nutrients in Runoff

By far the most prevalent nutrient found in the runoff from all sites that had dairy or beef cattle feedlot manure applied was Ca, ranging in concentrations from 4.1 to 673 mg L^{-1}, followed in decreasing order by Mg, K, S, Na, P, NO_3^--N, Mn, B, Fe, Zn, and Cu (Table 15, 16, and 17). Multiple simulations on the same plot affected nutrient concentrations in runoff similarly to the hydrological parameters. There was a clear reduction in the concentrations of all nutrients for each subsequent simulation. When the percentage change of each nutrient in the runoff of subsequent rainfall trials was averaged across all plots, a decline in concentrations was still present for all except Fe, Cu, Zn, and B. Two of the greatest reductions occurred with NO_3-N and dissolved P, which declined 44 and 20%, respectively, during the second trial (Table 18). This is not unexpected since nitrate is water soluble, and one would expect the greatest flush to occur in the first event. In the case of P, this is probably due the lack of time to.

Table 15. Means of nutrient concentrations in runoff using the first rainfall event averaged among three replicates.

Plot	NO_3^--N	B	Na	Mg	TP†	DP‡	S	K	Ca	Mn	Fe	Cu	Zn
	—	—	—	—	—	—	mg L^{-1}	—	—	—	—	—	—
A	0.34	0.19	4.9	8.9	7.4	2.80	1.1	14.7	220.6	0.69	0.32	0.01	0.05
B	0.34	0.23	8.4	31.1	9.3	5.26	2.0	29.7	673.0	1.10	0.08	0.01	0.02
C	0.29	1.16	5.6	4.3	5.2	3.30	2.3	28.7	11.6	0.33	0.39	0.03	0.03
D	1.40	0.12	5.5	6.1	4.1	2.57	0.9	10.9	19.2	0.89	1.27	0.03	0.08
E	0.85	1.73	5.8	5.1	3.2	2.96	1.0	15.5	26.4	0.80	0.62	0.03	0.09
F	2.92	0.11	5.2	8.4	3.9	4.88	1.4	27.2	56.9	1.63	0.98	0.06	0.21
G	1.10	0.07	4.2	3.8	3.1	2.61	0.5	17.0	27.6	0.34	0.77	0.05	0.15
H	3.15	0.13	17.0	3.1	3.8	3.81	2.3	13.7	9.1	0.02	0.10	0.10	0.03
I	1.82	0.16	7.6	3.7	2.4	2.42	1.6	8.2	25.5	0.04	0.08	0.04	0.03
J	1.71	0.18	9.3	4.0	1.5	0.39	0.7	11.6	17.1	0.52	0.11	0.03	0.02
K	0.60	0.12	7.9	6.8	3.5	2.78	5.7	24.0	36.5	0.25	0.32	0.03	0.06
L	5.19	0.08	17.1	9.1	6.6	4.55	1.7	26.6	324.7	0.53	0.29	0.05	0.10
M	1.00	0.02	3.0	3.1	0.9	0.62	3.8	5.1	18.7	0.23	0.79	0.06	0.06
N	0.94	0.02	3.2	3.4	1.8	1.67	2.9	5.2	12.4	0.10	0.40	0.09	0.04
O	1.29	0.14	2.6	2.6	2.7	2.28	0.8	6.7	8.0	0.02	0.06	0.03	0.03
P	2.37	0.03	0.6	2.7	1.8	1.83	0.9	5.9	12.0	0.03	0.05	0.01	0.04
Q	2.07	0.00	1.3	1.9	3.1	1.12	2.1	2.1	18.2	0.16	0.39	0.02	0.06
R	0.26	0.07	7.0	1.5	1.2	1.09	5.7	3.1	4.9	0.06	0.09	0.04	0.03
S	0.52	0.04	2.0	2.1	1.7	1.21	0.9	4.0	7.3	0.13	0.11	0.02	0.04
T	0.41	0.02	2.4	1.2	1.3	1.06	2.3	2.0	5.3	0.07	0.08	0.01	0.03
U	0.20	0.03	4.6	2.1	1.0	0.40	3.5	1.7	8.5	0.22	0.29	0.01	0.02
V	0.77	0.10	11.8	12.6	7.2	4.94	17.3	23.9	356.8	1.13	0.07	0.04	0.12
W	1.80	0.07	8.4	12.7	8.9	6.15	8.5	24.5	162.9	3.05	0.13	0.01	0.11

† TP, total P.
‡ DP, dissolved P.

Table 16. Means of nutrient concentrations in runoff using the second rainfall event averaged among three replicates.

Plot	NO_3^--N	B	Na	Mg	TP†	DP‡	S	K	Ca	Mn	Fe	Cu	Zn
	—	—	—	—	—	—	mg L^{-1}	—	—	—	—	—	—
A	0.07	0.61	4.0	7.6	4.15	2.37	0.6	10.4	179.1	0.56	0.36	0.02	0.05
B	0.09	0.63	5.1	23.2	6.89	4.64	1.1	19.0	487.5	1.11	0.22	0.01	0.04
C	0.12	1.16	3.6	3.3	3.09	2.02	1.0	13.9	9.5	0.29	0.37	0.03	0.03
D	1.31	0.14	4.4	5.1	3.84	2.19	0.4	8.8	16.9	0.82	1.12	0.03	0.06
E	0.98	1.51	4.5	5.6	2.27	2.84	0.6	15.6	28.3	0.84	0.68	0.03	0.08
F	0.80	0.08	4.3	4.7	4.06	2.69	0.5	14.6	30.8	0.86	0.74	0.05	0.13
G	0.46	0.05	4.0	2.7	2.12	1.66	0.3	11.0	18.4	0.21	0.63	0.05	0.11
H	2.49	0.13	17.6	2.8	2.98	3.00	1.3	8.7	8.9	0.03	0.14	0.11	0.03
I	0.90	0.27	6.1	3.7	2.05	2.07	0.8	3.0	23.5	0.04	0.07	0.06	0.03
J	0.93	0.17	10.4	2.5	0.87	0.25	0.4	3.1	11.3	0.35	0.14	0.07	0.02
K	0.19	0.10	7.0	4.8	2.32	2.13	2.4	12.1	25.7	0.15	0.21	0.03	0.03
L	1.76	0.07	15.1	6.6	4.60	3.85	0.9	16.2	264.5	0.49	0.41	0.05	0.10
M	0.32	0.00	1.8	2.0	0.48	0.35	3.0	2.5	11.4	0.13	0.45	0.04	0.04
N	0.59	0.02	2.2	3.1	1.80	1.53	3.1	3.6	12.8	0.11	0.42	0.09	0.04
O	0.53	0.19	1.1	2.8	2.24	2.08	0.3	2.6	8.9	0.01	0.03	0.02	0.02
P	0.72	0.01	0.9	1.5	1.57	0.93	0.2	2.3	7.8	0.03	0.03	0.01	0.03
Q	1.19	0.00	0.9	1.3	1.84	0.92	1.4	1.8	15.4	0.15	0.31	0.02	0.05
R	0.25	0.34	7.1	1.9	1.13	1.03	7.8	2.6	5.5	0.04	0.07	0.04	0.02
S	0.47	0.04	2.3	1.6	1.17	0.94	1.8	2.7	5.4	0.08	0.06	0.01	0.03
T	0.31	0.02	2.2	1.2	1.12	1.01	2.1	1.8	5.1	0.06	0.07	0.01	0.02
U	0.17	0.02	4.5	1.4	0.55	0.28	4.0	1.2	5.9	0.15	0.18	0.01	0.02
V	0.64	0.09	12.3	12.3	8.62	5.00	16.3	23.4	332.8	1.12	0.08	0.04	0.11
W	0.79	0.04	4.8	8.7	5.82	4.21	7.0	15.6	111.2	2.13	0.13	0.01	0.08

† TP, total P.
‡ DP, dissolved P.

Table 17. Means of nutrient concentrations in runoff using the third rainfall event averaged among three replicates.

Plot	NO_3^--N	B	Na	Mg	TP†	DP‡	S	K	Ca	Mn	Fe	Cu	Zn
	—	—	—	—	—	—	mg L^{-1}	—	—	—	—	—	—
A	0.06	0.87	3.0	5.3	2.77	1.66	0.3	7.4	116.2	0.37	0.32	0.03	0.03
B	0.03	0.87	4.1	21.5	4.16	3.85	0.8	16.3	396.1	1.04	0.38	0.01	0.05
C	0.13	1.24	3.3	3.1	3.06	1.83	0.7	10.6	9.7	0.33	0.42	0.03	0.03
D	0.67	0.20	4.0	4.7	3.59	1.99	0.2	7.6	15.3	0.75	1.03	0.03	0.08
E	0.84	1.70	3.8	4.4	1.87	2.30	0.4	12.1	21.6	0.58	0.59	0.05	0.07
F	0.43	0.06	3.9	3.7	3.14	2.11	0.3	9.8	25.6	0.65	0.61	0.05	0.13
G	0.25	0.04	4.0	2.1	1.48	1.22	0.2	7.2	13.7	0.16	0.51	0.04	0.07
H	1.38	0.15	17.2	2.3	2.41	2.40	0.8	6.4	8.0	0.02	0.10	0.11	0.03
I	2.62	0.30	6.1	3.1	1.49	1.56	0.6	3.0	17.4	0.02	0.04	0.05	0.02
J	0.36	0.17	9.7	2.4	0.67	0.23	0.5	5.5	11.4	0.36	0.17	0.11	0.03
K	0.13	0.09	6.5	3.4	1.63	1.47	1.2	7.4	20.6	0.10	0.12	0.03	0.02
L	1.01	0.05	16.5	5.5	3.85	3.30	0.7	11.7	228.5	0.42	0.37	0.06	0.09
M	0.30	0.00	1.8	1.7	0.44	0.30	2.5	2.2	11.1	0.13	0.41	0.04	0.05
N	0.40	0.07	2.0	2.7	1.76	1.31	2.2	2.7	11.0	0.08	0.30	0.07	0.04
O	0.38	0.15	0.8	2.4	2.12	1.71	0.2	2.3	8.8	0.01	0.05	0.02	0.02
P	0.49	0.00	0.2	1.2	0.83	0.65	0.1	1.8	6.3	0.02	0.02	0.00	0.02
Q	0.83	0.00	0.9	1.3	1.42	0.95	1.5	1.9	14.0	0.12	0.34	0.02	0.06
R	0.23	0.26	9.6	2.9	1.03	1.24	11.5	2.6	8.1	0.04	0.10	0.06	0.03
S	0.21	0.03	2.6	1.5	1.16	0.90	1.6	2.5	5.1	0.07	0.06	0.02	0.03
T	0.27	0.01	1.9	1.0	0.90	0.79	1.8	1.5	4.1	0.05	0.05	0.01	0.02
U	0.12	0.02	4.8	1.2	0.46	0.24	5.2	1.1	5.1	0.13	0.13	0.01	0.01
V	0.43	0.11	13.8	16.1	8.25	5.55	18.5	33.0	443.6	1.10	0.06	0.04	0.09
W	0.45	0.03	3.9	6.6	4.54	3.11	5.8	11.7	83.9	1.52	0.12	0.01	0.07

† TP, total P.

‡ DP, dissolved P.

Table 18. Percent decline in nutrient concentration due to subsequent rainfall events.

First to Second Rainfall Event				Second to Third Rainfall Event			
Element or ion	% Change			Element or ion	% Change		
	mean	sd	cv		mean	sd	cv
NO_3^-	-44.3	26.5	59.8	NO_3^-	-54.4	31.5	58.0
K	-33.2	18.6	56.0	K	-41.4	25.5	61.6
S	-29.3	31.3	106.7	S	-32.5	24.0	73.9
T.P.	-20.2	21.1	104.4	T.P.	-31.9	20.4	64.0
D.P	-20.1	14.7	73.5	D.P	-31.3	19.8	63.2
Mn	-18.9	20.2	107.1	Mn	-26.0	82.8	318.7
Mg	-17.5	18.6	106.2	Mg	-24.1	32.4	134.7
Ca	-16.9	17.4	102.5	Ca	-22.9	28.7	125.2
Na	-1.3	80.8	6410.4	Na	-7.4	54.4	734.6
Cu	15.7	69.5	443.9	Cu	54.0	292.7	541.8
Fe	29.8	143.1	480.3	Fe	79.8	325.2	407.5
Zn	75.1	297.8	396.5	Zn	354.5	1656.3	467.2
B	259.1	1243.1	479.8	B	633.3	2945.5	465.1

replenish the easily sorbed P between rainfall events. The decline in nutrient concentration tended to be greater during the third event than the second (Table 18), with NO_3-N, K, P, and Mn having the greatest decline. The micronutrients, Fe, Cu, Zn, and B, measured concentrations were usually so low that no significant change could be measured in most cases for these elements. One reason for the lower variability in the determination of DP concentration using multiple rainfall events is because there was very little change in concentrations between the second and third rainfall events, most of the difference occurred after the first simulation. The sharp decline in DP is likely due to the depletion of the easily desorbed P between the first and second run. Averaging nutrient concentrations for multiple runs on a plot would result in the underestimation of nutrient loads (losses) in the runoff. For example DP and TP concentrations for subsequent runs declined by 26 and 27%, respectively from the first simulation. Total runoff volume on the other hand increased by 26%, while DP and TP loads were reduced only 6 and 8 %.

Just as with the hydrological parameters, the pooling of all runs together for statistical purposes increases the variability of all measured nutrients (Table 19 a and b). In the case of TP and DP, we were underestimating TP and DP loads (total losses) by an average of 27 and 26%, respectively, using multiple rainfall runs on each plot. This was the basis for using only the first simulation in calculating losses to be used for evaluation of the P Index. This trend was present in our studies even when draining period was increased to 24 hours. Previous studies in chapter III did not show a significant change, however the draining period in those trials was 40 hrs. Due to logistical reasons we were

Table 19a. Coefficients of variability for nutrient concentration using multiple rainfall events on each plot.

Plot	NO_3^--N cv	B cv	Na cv	Mg cv	D.P. cv	S cv	K cv	Ca cv	Mn cv	Fe cv	Cu cv	Zn cv
A	91	141	25	33	32	53	32	44	38	15	83	20
B	116	130	42	19	26	47	35	29	18	71	37	58
C	69	20	30	24	41	64	51	18	20	12	29	26
D	80	61	24	24	18	70	22	29	29	28	28	37
E	48	23	27	20	20	45	22	20	28	14	44	27
F	55	15	6	13	14	42	14	18	31	7	13	24
G	8	7	1	16	26	17	2	32	22	23	8	28
H	22	9	9	9	16	40	22	10	11	11	6	14
I	53	7	12	12	13	24	31	30	25	8	13	14
J	32	9	43	27	30	22	57	25	26	13	15	26
K	52	10	23	5	22	23	32	17	4	25	8	30
L	29	2	22	19	5	17	20	6	6	14	29	16
M	39	33	4	4	9	16	14	19	43	39	13	12
N	17	48	8	4	13	8	18	9	31	37	14	11
O	5	22	33	9	10	11	33	2	65	67	19	20
P	52	45	33	18	22	34	17	33	40	26	21	23
Q	8	0	43	47	36	16	26	29	50	29	11	30
R	17	10	53	28	30	46	13	28	28	29	36	17
S	41	33	31	48	70	56	34	37	4	14	10	27
T	18	6	12	15	36	18	10	16	83	49	3	18
U	42	68	31	19	13	40	6	24	14	13	78	82
V	34	14	14	20	12	10	28	20	3	22	14	15
W	25	19	3	13	11	2	6	11	14	15	4	16

Table 19b. Coefficients of variability for nutrient concentration using the first rainfall event on each plot.

Plot	NO_3^--N cv	B cv	Na cv	Mg cv	D.P. cv	S cv	K cv	Ca cv	Mn cv	Fe cv	Cu cv	Zn cv
A	22	48	9	34	30	9	17	43	36	3	39	15
B	61	28	31	4	13	18	21	10	16	134	25	145
C	54	27	16	16	33	34	22	5	17	13	30	16
D	76	5	21	13	12	43	17	2	6	42	34	25
E	28	23	20	18	18	27	14	19	26	14	42	42
F	59	10	10	20	17	46	15	23	43	9	13	26
G	13	14	3	21	33	9	13	39	27	26	11	25
H	21	9	10	21	27	64	40	27	31	30	7	24
I	11	3	33	16	13	13	39	23	15	6	64	9
J	29	17	60	42	47	57	74	41	39	21	123	43
K	70	19	51	9	15	28	38	25	5	30	30	40
L	36	14	28	17	17	21	24	8	29	74	44	84
M	47	33	28	1	7	14	16	13	32	31	7	44
N	16	72	6	4	15	8	20	8	26	32	16	14
O	8	11	17	14	13	15	38	3	71	84	14	29
P	54	55	119	23	22	39	21	33	43	19	33	14
Q	24	0	42	52	37	22	30	36	54	27	20	37
R	28	90	74	27	32	73	17	30	42	38	52	15
S	72	54	50	53	68	0	44	38	3	13	23	34
T	42	8	16	14	35	23	12	12	65	39	5	23
U	35	62	26	15	9	28	5	24	13	19	59	63
V	52	19	13	25	26	14	28	28	23	20	21	30
W	30	20	6	17	14	3	6	15	15	8	7	18

unable to wait 40 hours between runs. Also, many of the fields contained livestock which were attracted by our frames and would trample and deposit fresh manure on plots overnight. We decided that a shorter draining period would yield better results than having animals disturb our plots. The high level of variability for the pooling of all runs was our justification for using only the first event for the rest of our analyses. However, this still provides three replicates for each of twenty-three sites scattered across the High Plains and Central regions of Texas where most CAFOs currently are located. An argument could be made that inclusion of subsequent events better reflect the overall P losses to be expected from a field during the season. The use of multiple runs might be a better indication of yearly losses, but the lack of time between simulations fails to allow soluble P levels to equilibrate with other forms in the soil. The use of three events would probably be an underestimation of total P losses. All future analyses discussed will only include the first simulation event.

Effects of Management

The average time of rainfall needed to generate 30-minutes of continuous runoff ranged from 2.6 to 37 minutes, the total applied amount of rainfall needed was 40.1 to 74.5 mm, runoff depth ranged from 17.1 to 50.6 mm, and total sediment with particulate matter increased from not measurable to 1173 g (Table 14). Statistical analysis showed that rainfall needed to generate runoff and total sediment and particulate matter lost had the greatest variability among all plots. Runoff amount and total rainfall were highly reproducible. This indicates that initial application of rainfall and sediment losses are hard to determine probably as a result of differences in surface roughness, cracks,

vegetation type, and management practices. While slope is one of the primary factors influencing soil erosion, this relationship was not significant (significance declared as $p \leq 0.1$) for a simple linear regression in predicting total particulate matter lost (Table 20). Regression analysis showed that slope alone was not a significant factor in determining the amount of rainfall needed to generate continuous runoff, runoff volume depth, total rainfall applied, or sediment and particulate matter lost from plots. The greatest impact was due to the type of vegetative cover. On a bare soil regardless of texture, an increase in slope results in reduction of the amount of rainfall needed to generate a continuous 30 min runoff. This is to be expected as the lack of vegetation reduces the infiltration rate and the amount of water that can be stored by leaves before runoff can occur. The trend could not be confirmed in soils with vegetative cover ($p \leq 0.1$) for either rainfall applied, runoff volume, total rainfall applied, or particulate matter lost, although relatively high r^2 (0.39 to 0.65) values were obtained for sites with only row crops. Correlations could be improved (higher r^2) through the grouping of sites by type of vegetative cover (bare, row crop, and bermudagrass) and texture (sand or clays). In all cases increasing slopes resulted in negative responses to the amounts of rainfall needed to generate runoff and total rainfall applied while increasing the amount of particulate matter lost. However, while increasing slopes generated greater volumes of runoff in fine textured soils, we could not reproduce these results with coarse textured ones. In fact, we obtain a negative correlation on sandy sites with bermudagrass ($r^2=0.29$; $p=0.27$). This unusual behavior was an anomaly created by the inability to completely seal openings around the edges of our metal frames. This problem was not present with sticky clayey soils, but was

Table 20. Coefficient of determination for slope against rainfall, runoff, total rainfall application, and particulate matter lost.

	Rainfall for runoff	Level of significance	Runoff volume	Level of significance	Total rainfall	Level of significance	Total sediment	Level of significance
	r^2	p	r^2	p	r^2	p	r^2	p
All	0.001	0.89	0.002	0.86	0.001	0.89	0.04	0.36
Clay	0.01	0.70	0.09	0.29	0.01	0.70	0.05	0.45
Sand	0.02	0.75	0.06	0.54	0.02	0.75	0.38	0.08
Bare	0.28	0.14	0.15	0.30	0.28	0.14	0.43	0.06
Row crop	0.65	0.19	0.55	0.26	0.65	0.19	0.37	0.39
Bermudagrass/ row crop	0.06	0.49	0.00	0.92	0.06	0.49	0.01	0.74
Clay*bare	0.27	0.29	0.36	0.21	0.27	0.29	0.84	0.01
Clay*row crop	0.65	0.19	0.55	0.26	0.62	0.19	0.37	0.39
Clay*bermudagrass	0.47	0.32	0.80	0.10	0.47	0.32	0.63	0.20
Sand*bare	0.71	0.36	0.05	0.86	0.71	0.36	0.95	0.15
Sand*bermudagrass	0.03	0.76	0.29	0.27	0.03	0.76	0.14	0.46

exacerbated by increasing slopes in sandy sites. The overall result was a decrease in our recorded volumes due to significant leaks at the edges of the frames that prevented the total accumulation of the generated runoff in our flumes.

Figure 12 illustrates the effects of cover on applied rainfall, runoff volume, total rainfall applied, and particulate matter lost by plots. Grass covered plots compared to bare ground increased the amount of rainfall needed to generate runoff from 6.2 to 14.2 mm, and total rainfall applied for a 30 minute event from 43.7 to 51.7 mm, while decreasing runoff amount from 42.2 to 28.8 mm and drastically decreasing particulate matter from 502 to 37.8 g. While hydrological parameters for bermudagrass covered sites were statistically different relative to bare cover and row crop sites, the difference was not significant between row crop and bare soils. Dissolved P and TP concentrations, loads, and percent DP were influenced by the management practice of vegetative cover. Runoff from bare soils had the highest concentrations of DP and TP at 3.3 and 5.5 mg L^{-1}, respectively. Row crop generated DP and TP concentrations of 2.6 and 3.1 mg L^{-1}, while bermudagrass covered sites had the lowest at 2.1 and 2.5 mg L^{-1}, respectively (Table 21). Dissolved P and TP loads followed the same trend. The DP portion of TP ranged from 27% to 100%. The highest percentage was from sites with grass covers, 86%, followed by row crops with 75%, and bare soil with 60%. A reason for this trend is the greater amounts of particulate matter in the runoff of soils. While the percentage of DP is higher on grass covered plots, the DP load is less than half that recorded for the bare ground sites, and less than one third for TP losses. It appears obvious that management practices must be considered, and can have a greater effect on

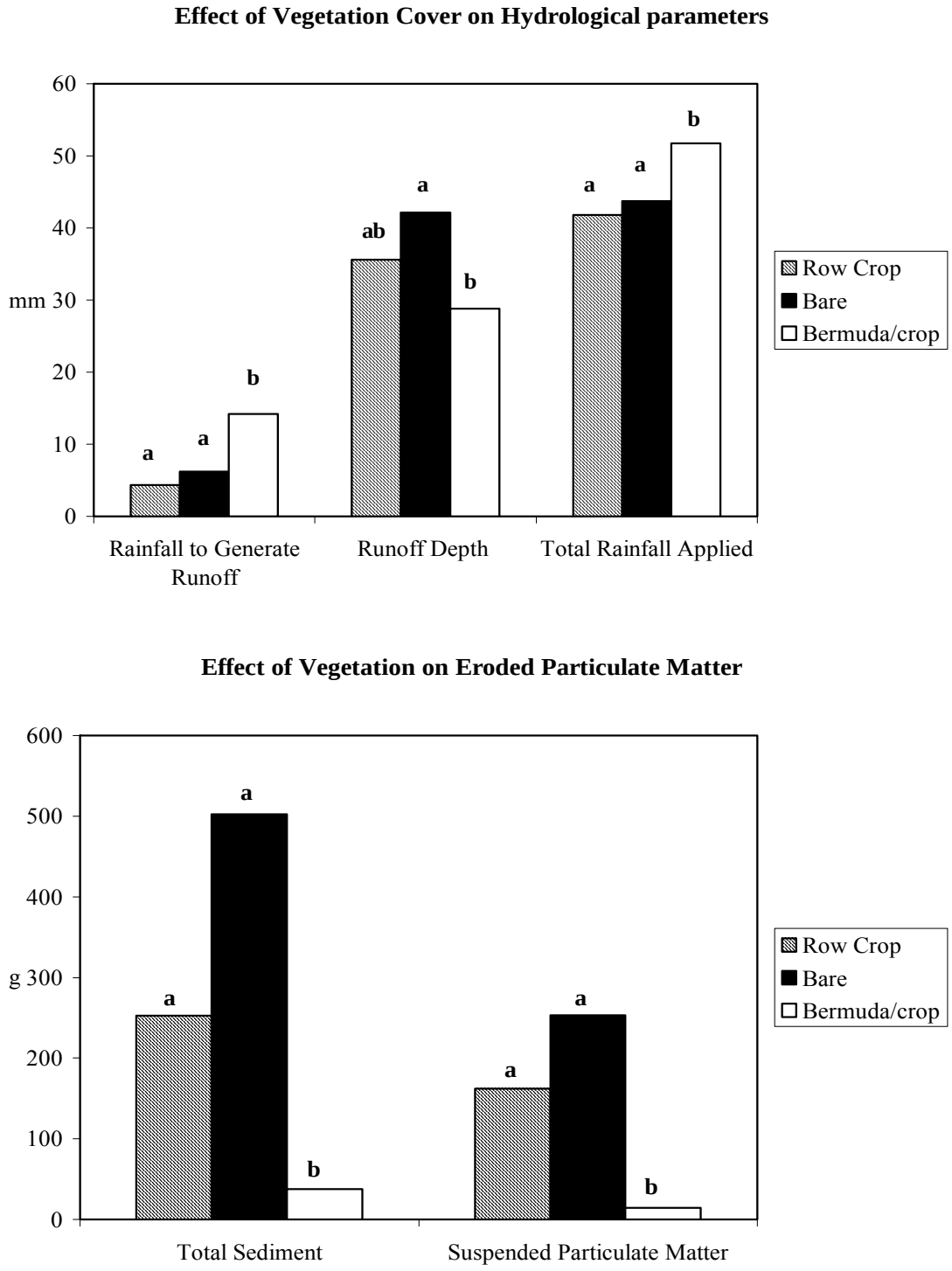

Fig. 12. Effects of management practice on hydrological parameters and particulate matter lost.

Table 21. Effects of management on hydrological parameters, total and dissolved phosphorus.

	Time		Rainfall for runoff	Total runoff	Runoff depth	Total rainfall	Total sediment
	secs.	min	mm	L	mm	mm	g
Bare	298	5.0	6.2	126	42.2	43.7	502.0
Crop	207	3.4	4.3	107	35.6	41.8	252.4
Bermuda/crop	682	11.4	14.2	86	28.8	51.7	37.8

				Reduction of P concentration by cover	
	DP	TP	DP	DP	TP
	mg L^{-1}	mg L^{-1}	%	%	%
Bare	3.3	5.4	61	—	—
Crop	2.6	3.1	83	20	42
Bermuda/crop	2.1	2.5	85	36	54

	DP load	TP load	DP	Reduction of load by cover	
	mg P	mg P	%	% DP	% TP
Bare	417	693	60	—	—
Crop	270	325	83	35	53
Bermuda/crop	173	202	85	59	71

losses than physical characteristics alone. The overall conclusion is that a good ground cover of grass, such as common or Coastal (hybrid) bermudagrass, is essential in increasing the amount of soil infiltration, thereby reducing the volume of runoff and sediment load.

As time of rainfall application progressed, concentrations of all nutrients in collected runoff declined except for Fe and Zn. This was evident by the decrease in measured conductivity of the runoff during the course of the simulations (Appendix Table A-2). As mentioned before, the low concentrations of Fe, Cu, Zn, and B in most plots resulted in a large variability among sites, making it impossible to establish any general trends. Concentration of DP from sites that had manure applied were in excess of plant nutrients necessary to meet nutritional needs ranging from 0.39 to 6.15 mg L^{-1} (Table 15). Statistical analysis showed that DP and TP concentrations 15-minutes after runoff started were significantly greater than at the 30-minute period in 18 of the 23 sites for DP and in 14 of 23 sites for TP (Table 22). This would seem to indicate that greater losses would be expected in the first few minutes from short, high-intensity storms, probably from the first storm event in the year as proposed by Pote et al. (1996). Reduction in the concentrations of DP and TP were proportional so that the portion of DP remained constant within sites for the duration of the simulation, which ranged in our study from 27 to 100% of total suspended P in runoff. Neither pH nor soil texture affected percent DP in runoff, only management practice in the form of ground cover could be used to separate levels (Table 21).

Table 22. Paired t-test level of significance for differences in dissolved (DP) and total P (TP) concentration due to time.

Plot	DP 1-tail (p)	TP 1-tail (p)
A	0.023	0.030
B	0.204	0.005
C	0.001	0.000
D	0.079	0.045
E	0.012	0.095
F	0.015	0.105
G	0.008	0.040
H	0.001	0.170
I	0.097	0.045
J	0.365	0.175
K	0.000	0.000
L	0.142	0.095
M	0.026	0.060
N	0.002	0.015
O	0.371	0.035
P	0.001	0.125
Q	0.018	0.215
R	0.004	0.010
S	0.001	0.005
T	0.071	0.025
U	0.341	0.255
V	0.001	0.355
W	0.001	0.000

Extractable Plant Nutrient in Soil

Extractable plant nutrient concentrations in soils using the TAMU extractant, as well as P using Mehlich III and CSSP for depths of 0-5, 5-15, and 0-15 cm are listed in Tables 23, 24, and 25. Concentrations for extractable nutrients were evenly distributed among the three soil depths used for all except extractable P. This was not surprising as P is the most insoluble of the elements measured and tends to accumulate at the soil surface, despite the fact that most of the sites had manure effluent applied and were tilled on a regular basis. In the case of P, the percentage extractable P in the 0-5 cm portion of the soil profile ranged from a low of 37 to 92% of the total extracted for a depth of 0-15 cm. Using ICP techniques, TAMU solution consistently extracted more P in 48 of 69 plots than did Mehlich III, and both of these extracted much higher concentrations than the calcium solution extractant (CSSP). The TAMU method extracted P ranging from 35 to 2606, 12 to 1101, and 20 to 1607 mg kg^{-1}, for the top 5 cm, 5 to 15 cm, and top 15 cm soil depth, respectively. Mehlich III extracted higher amounts of P mostly in plots with the lower soil P contents (less than 200 mg P/kg soil). As soil P content increased the extracting efficiency of the TAMU method also increased and eventually surpassed Mehlich III. At the highest P content, the TAMU extraction removed more than twice as much P as Mehlich III. The $CaCl_2$ (CSSP) extraction yielded the lowest concentrations of extractable P in all cases.

The higher amounts of P removed by the TAMU method compared to the Mehlich III method would indicate that while a significant amount of P present is bound to amorphous oxides of Al and Fe, there is also a significant portion associated with Ca

Table 23. Extractable soil nutrients in 0-5 cm depth.

PLOT	CSSP†	Mehlich III-P	TAMU-P‡	Na	Mg	S	K	Ca	B
	—	—	—	mg kg^{-1}	—	—	—	—	—
A	10.22	227	451	499	1000	138	1354	38177	2.4
B	10.01	235	467	543	2200	175	1411	58996	2.5
C	11.26	325	339	513	782	97	1722	4456	3.3
D	20.04	846	608	533	866	186	1624	5426	3.4
E	15.73	701	625	579	719	181	1735	6766	3.7
F	16.06	664	1213	282	833	186	1333	13731	2.5
G	22.74	1221	2606	352	1199	248	2107	23022	4.2
H	7.42	565	602	331	978	118	701	23238	1.1
I	17.15	1098	1729	344	1584	256	806	181664	2.1
J	3.70	79	45	341	415	75	975	5952	0.5
K	4.74	108	89	270	350	80	613	40019	0.4
L	11.65	651	764	401	1428	257	1400	163277	2.1
M	13.99	120	213	281	367	130	249	22402	0.7
N	14.57	162	285	286	522	142	440	28424	0.8
O	49.63	485	527	249	245	121	183	9094	1.0
P	57.88	421	639	287	299	165	261	14049	0.9
Q	5.03	104	54	308	167	111	187	2440	0.1
R	16.16	661	771	358	750	150	1175	16652	1.8
S	9.82	306	315	275	290	93	546	4794	0.7
T	25.83	766	1202	322	777	220	346	16748	1.9
U	2.25	65	35	340	191	52	133	1636	0.2
V	5.26	266	359	442	1016	116	1458	69192	0.6
W	7.59	356	626	448	900	118	1742	29014	1.2

† CSSP, calcium solution soluble P.

‡ TAMU P, acidified ammonium acetate-EDTA extracted P.

Table 24. Extractable soil nutrients in 5-15 cm depth.

PLOT	CSSP†	Mehlich III-P	TAMU-P‡	Na	Mg	S	K	Ca	B
	—	—	—	mg kg^{-1}	—	—	—	—	—
A	7.77	155	364	556	992	174	1211	40497	2.3
B	7.15	158	373	603	2308	207	1392	63273	2.3
C	9.42	258	270	581	755	107	1365	4003	2.6
D	13.86	566	420	531	809	162	1463	4748	2.8
E	11.87	513	524	577	719	172	1464	7164	3.0
F	5.52	251	402	315	732	108	1237	10384	1.7
G	9.99	580	1101	357	978	155	2129	16711	2.7
H	2.03	99	62	346	752	57	571	6690	0.3
I	7.50	480	594	321	1264	183	705	211207	1.4
J	2.59	34	19	323	331	58	595	6804	0.3
K	2.45	32	24	305	357	80	521	86413	0.3
L	9.21	476	558	407	1384	257	1308	180209	1.7
M	4.28	42	87	290	336	84	150	33491	0.1
N	5.02	55	113	318	440	109	240	41426	0.3
O	24.91	280	267	243	106	56	84	5147	0.2
P	18.47	277	354	272	131	71	124	7605	0.2
Q	2.70	26	12	288	164	76	170	2523	0.1
R	9.94	434	463	414	544	105	1031	11849	0.9
S	3.27	137	129	293	248	58	377	3341	0.4
T	9.72	207	229	273	259	67	220	5055	0.6
U	0.79	31	16	340	156	51	110	1544	0.2
V	2.95	197	257	491	973	119	1499	72430	0.6
W	6.37	303	492	480	844	110	1560	28043	1.2

† CSSP, calcium solution soluble P.

‡ TAMU P, acidified ammonium acetate-EDTA extracted P.

Table 25. Extractable soil nutrients in 0-15 cm depth.

PLOT	CSSP†	Mehlich III-P	TAMU-P‡	Na	Mg	S	K	Ca	B
	—	—	—	mg kg^{-1}	—	—	—	—	—
A	8.66	185	408	536	1010	172	1308	41092	2.4
B	9.02	193	417	543	2309	204	1446	62895	2.3
C	9.89	299	303	532	761	107	1544	4136	2.6
D	13.10	615	453	543	805	162	1454	4820	3.1
E	11.17	427	493	583	685	174	1440	7455	3.0
F	7.93	397	657	321	780	135	1311	10742	2.1
G	13.81	772	1607	373	1071	183	2117	19318	3.1
H	2.68	232	204	339	800	77	593	10892	0.5
I	9.97	656	846	312	1314	203	772	198598	2.0
J	2.94	45	26	339	357	64	750	6718	0.5
K	2.93	46	38	277	355	81	515	74907	0.4
L	10.52	562	614	385	1396	260	1276	177199	1.8
M	6.74	64	125	280	351	106	174	31753	0.3
N	7.86	84	160	307	453	123	303	35883	0.3
O	31.19	344	353	253	145	73	126	5077	0.5
P	28.33	332	457	265	177	97	153	9997	0.5
Q	2.81	43	20	283	165	81	173	2475	0.1
R	12.63	521	560	375	600	118	1064	13003	1.3
S	5.08	200	193	281	257	74	422	3945	0.5
T	15.03	351	440	279	365	98	250	7445	0.9
U	1.15	40	21	352	166	52	111	1502	0.2
V	4.72	257	344	491	1022	127	1589	73824	0.7
W	6.51	329	544	473	878	113	1660	28833	1.2

† CSSP, calcium solution soluble P.

‡ TAMU P, acidified ammonium acetate-EDTA extracted P.

that increased as soil P content increased. Studies by Cole et al. (1953) first indicated that at high concentrations of P and $CaCO_3$ (Table 26) in soil, P compounds are precipitated rather than adsorbed and that whatever these forms are they are more soluble than hydroxyapatite or fluoroapatite. It was later shown that bond energy for precipitated P-forms required less energy than $CaCO_3$ bound P (Holford and Mattingly, 1975). It is possible that these compounds could be Mg-bound P of such forms as $Mg_3(NH_4)_2(HPO_4)_4 \cdot 8H_2O$ or $Mg_3(PO_4)_2 \cdot 22H_2O$. The longer extraction period used by the TAMU procedure and greater amounts of Mg-P compounds could explain the higher STP found in solution.

The extracting efficiency of all methods was not affected by depth of soil core used. Concentrations of STP were always highest at the 0-5 cm depth, because most manure solids or effluent was surface applied. The second highest values were obtained for the 0-15 cm cores due to dilution of the surface fraction. Analyses from 5-15 cm depths were the lowest since the fraction that contained the greatest amount, especially in fields that have free roaming livestock was not included. However, the middle core was still high regardless of texture, indicating that not all of the applied P remains in the top 5 cm, but rather a significant amount is moved down the profile, at least to a depth of 15 cm. This agrees with finding by Hansen et al. (2004) movement down to 65 cm with both solid-manure and lagoon-manure additions.

Total soil P content from a 0-15 cm soil depth was highly correlated to Mehlich III and TAMU extracted P at all depths (Fig. 13, 14, and 15). One plot had an extremely high total soil P content (3366 mg kg^{-1}) with respect to other sites, however the

Table 26 Total organic carbon (TOC), soil organic matter (SOM), calcium carbonate equivalent (CCE), soil inorganic carbon (SIC) and total carbon (TC) in plots.

Plot	pH	TOC	SOM	Calcite	Dolomite	CCE	SIC	TC
		-	-	-	g/kg	-	-	-
Q	4.86	11.1	19.0	0.0	0.0	0.0	0.0	19.0
U	5.52	6.7	12.0	0.0	0.0	0.0	0.0	12.0
P	6.33	17.7	31.0	15.0	1.0	16.0	1.9	32.9
O	6.36	28.7	50.0	10.0	0.0	10.0	1.2	51.2
J	6.78	21.1	36.0	1.0	1.0	2.0	0.0	36.0
T	6.83	23.1	40.0	9.0	1.0	10.0	1.2	41.2
S	6.87	13.4	23.0	0.0	1.0	1.0	0.1	23.1
F	6.92	31.4	54.0	8.0	2.0	10.0	1.2	55.2
G	6.97	39.6	68.0	23.0	5.0	28.0	3.4	71.4
D	7.03	18.9	33.0	2.0	0.0	2.0	0.2	33.2
H	7.03	19.6	34.0	10.0	3.0	13.0	1.6	35.6
I	7.10	38.7	67.0	358.0	0.0	358.0	43.0	110.0
R	7.12	19.9	34.0	16.0	4.0	20.0	2.4	36.4
E	7.26	21.8	38.0	7.0	1.0	8.0	1.0	39.0
C	7.34	18.2	31.0	1.0	1.0	2.0	0.2	31.2
N	7.51	16.9	29.0	71.0	5.0	76.0	9.1	38.1
M	7.51	13.7	24.0	0.0	0.0	0.0	0.0	31.3
K	7.58	14.8	25.0	126.0	9.0	136.0	16.3	41.3
L	7.80	25.2	44.0	408.0	1.0	409.0	48.0	92.0
A	7.84	20.8	36.0	85.0	4.0	89.0	10.7	46.7
B	7.92	25.9	45.0	0.0	0.0	0.0	0.0	66.4
W	8.10	16.4	28.0	56.0	12.0	69.0	8.3	36.3
V	8.57	12.3	21.0	150.0	18.0	170.0	20.4	41.4

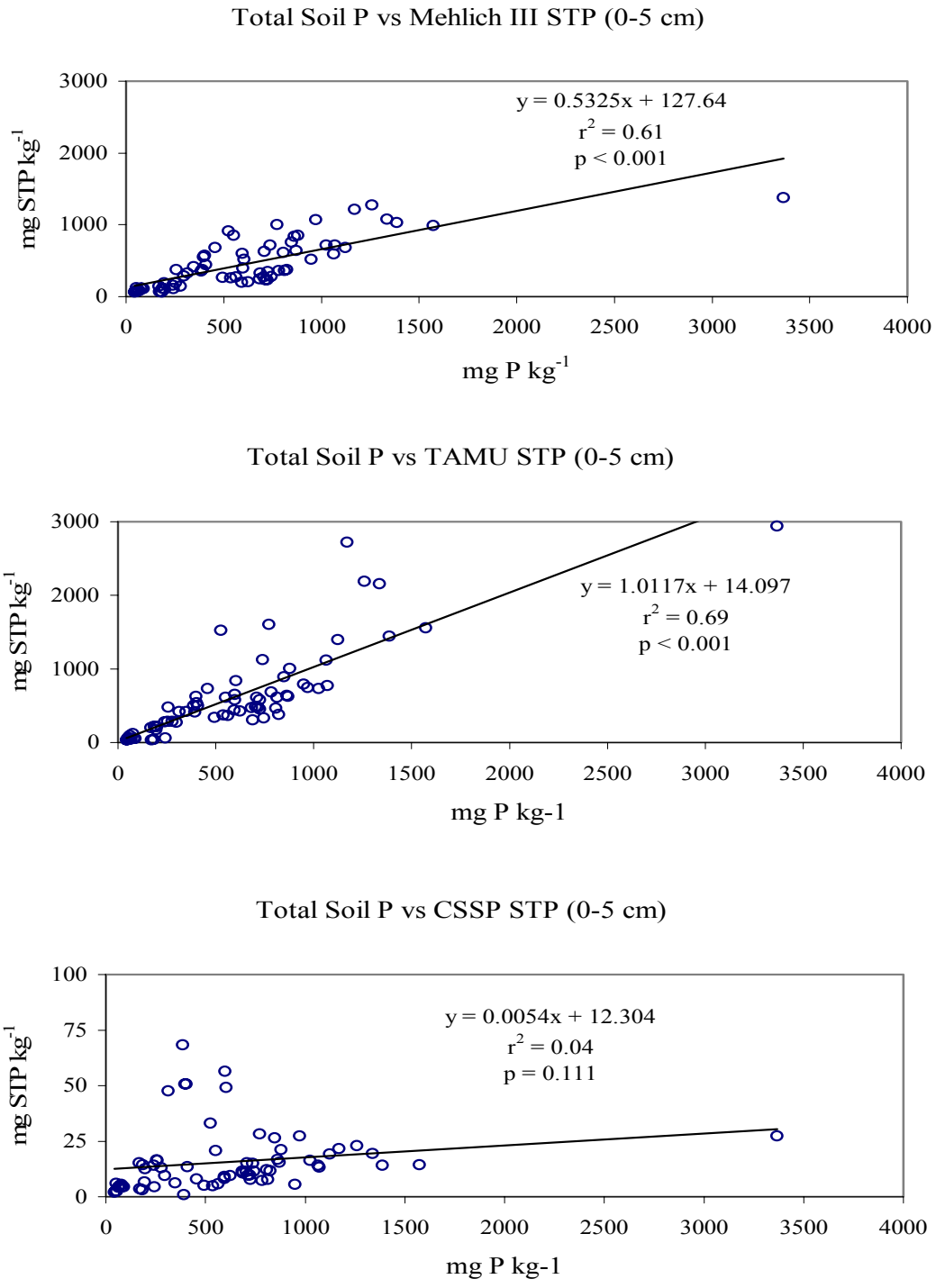

Fig. 13. Relationship between total soil P and soil test P (STP) for Mehlich III, ammonium acetate-EDTA (TAMU), and calcium solution soluble P (CSSP) at a 0-5 cm depth.

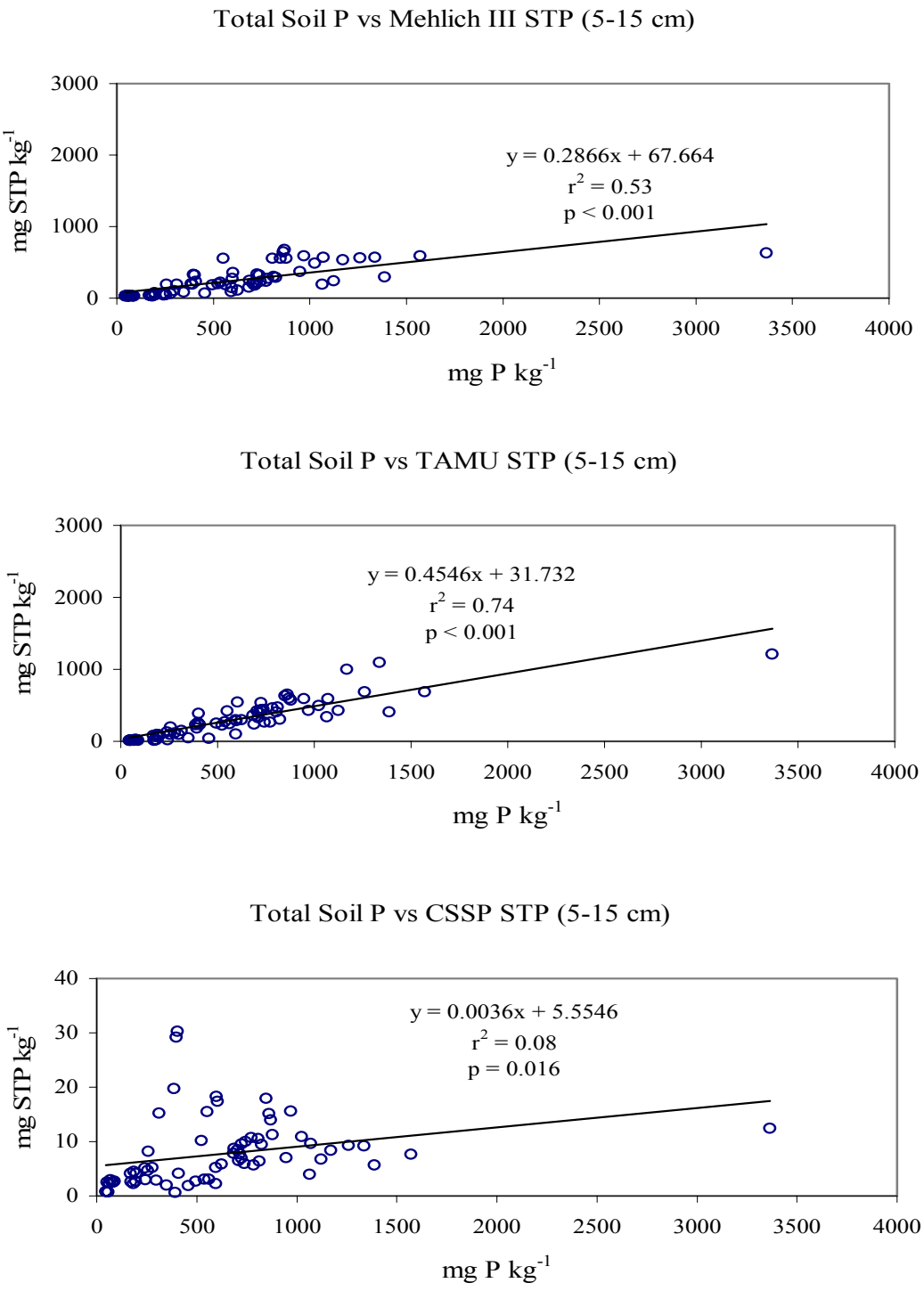

Fig. 14. Relationship between total soil P content and soil test P (STP) for Mehlich III, ammonium acetate-EDTA (TAMU), and calcium solution soluble P (CSSP) at a 5-15 cm depth.

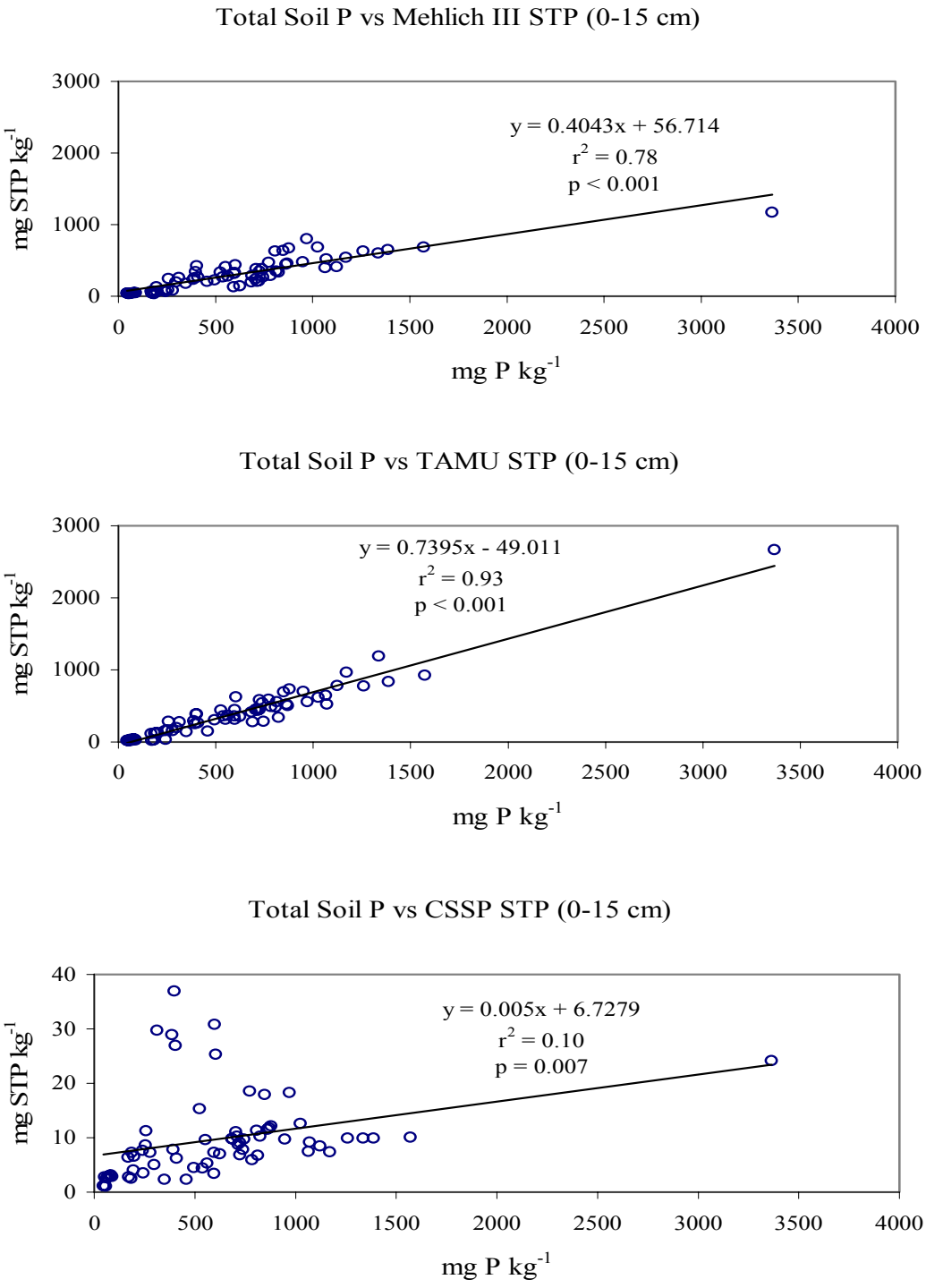

Fig. 15. Relationship between total soil P content and soil test P (STP) for Mehlich III, ammonium acetate-EDTA (TAMU), and calcium solution soluble P (CSSP) at a 0-15 cm depth.

exclusion of this point in regressions did not affect overall relationship (r^2) or calculated slopes but it did alter the x-intercepts. The slopes from Mehlich III were half those of the TAMU method at all depths. This result is expected since TAMU STP values were higher in 17 of the 23 sites used. The correlation between total soil P and STP, while significant for CSSP was very poor, with less than 10% of the variability accounted for by first order linear regression at a depth of 0-15 cm. One possible explanation is that the high levels of $CaCl_2$ solution (14.7 g L^{-1}) could be interfering in the extraction of P. Although many of the soils used were highly calcareous, the highest measured concentration of calcium in runoff was only 0.67 g L^{-1} (Tables 15, 16, and 17). Comparisons of P desorption rates between poultry and cattle amended soils have shown that P release to be inversely related to total Ca content, exchangeable Ca (Siddique and Robinson, 2004). This is attributed to the increase P sorption strength through the surface saturation of alumino-silicate clays with Ca-P complexes.

As expected, due to the many different factors affecting P sorption, there was no significant relationship between total soil P and runoff concentrations of either DP or TP (r^2 = 0.06, 0.16) with levels of significance at $p \geq 0.1$. However, when soils were separated by total soil P content below 700 mg kg^{-1}, a highly significant positive relationship ($p \leq 0.01$) for both DP and TP concentrations in runoff was found, r^2 of 0.43 and 0.41, respectively. Although not statistically significant due to a great deal of variability ($p \geq 0.5$), a negative correlation was apparent as soil P content increased above 700 mg kg^{-1} (Fig. 16). It would seem that solubility of all forms of P is reduced above a certain point of total soil P, which results in lower losses of suspended P than would be

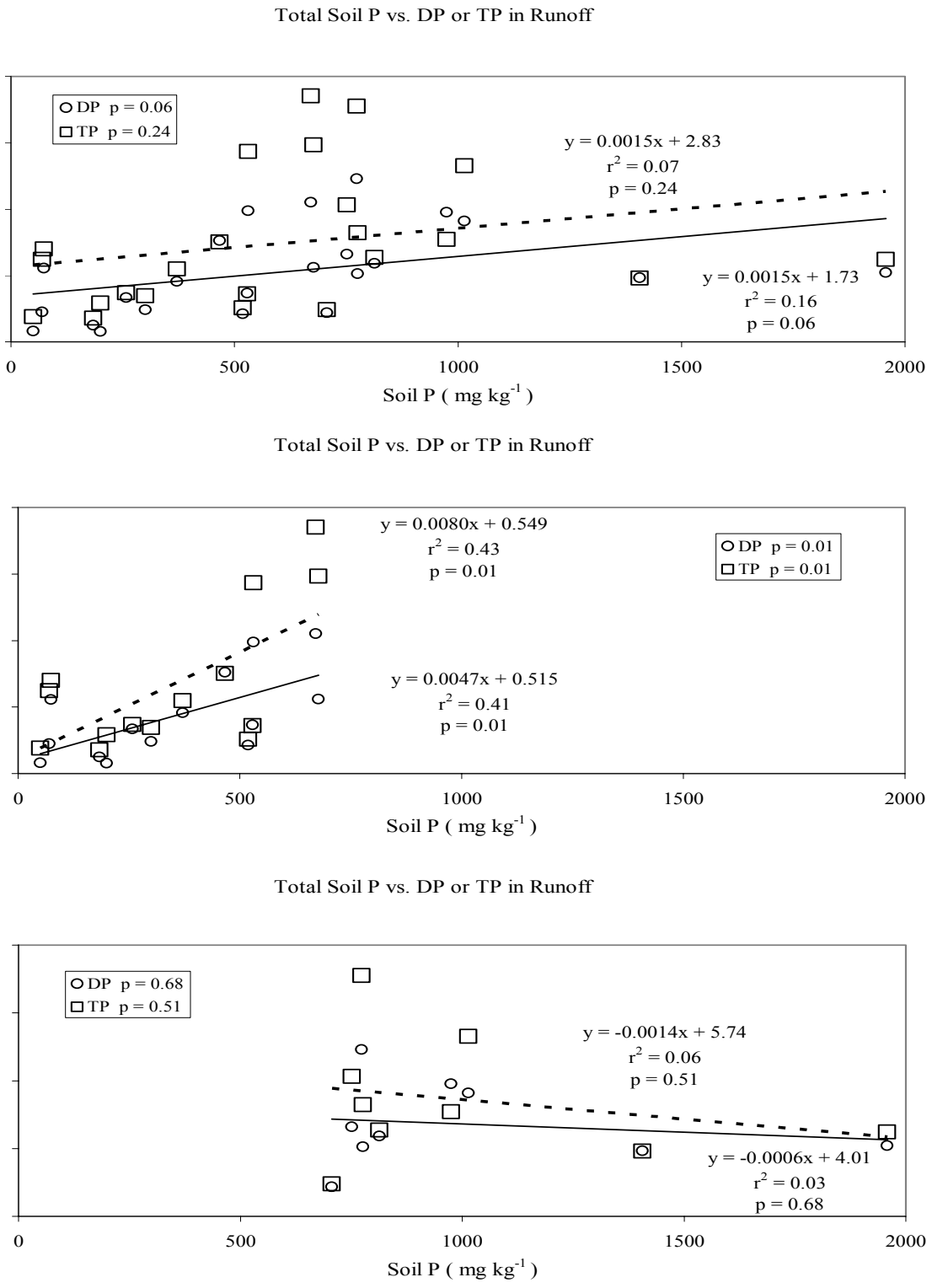

Fig. 16. Relationships between total soil P content and concentrations of P (dissolved and total P) released in runoff.

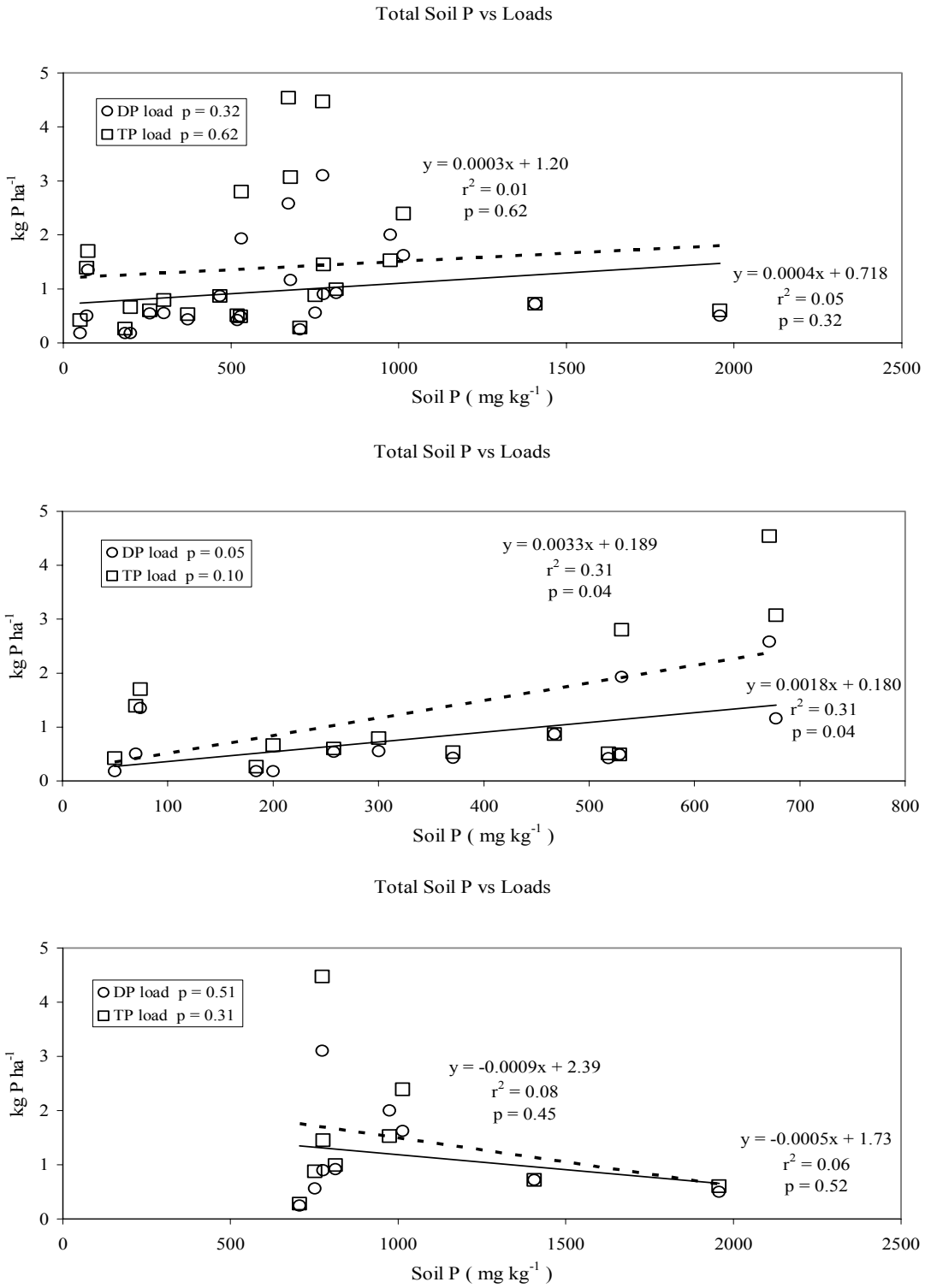

Fig. 17. Relationships between total soil P content and P loads (dissolved and total losses) due to runoff.

expected from such high STP values. This trend was also present when concentration of DP was plotted against the DP and TP loads (Fig. 17). They all provide further evidence that either some sparingly soluble calcium-bound P form is precipitating in soils, or P sorption strength is being increased by formation of a surface complex between Ca and P or by the binding of exchangeable Ca to hydrous oxides in clay minerals. While these P compound might still be extracted by both the strongly acidic Mehlich III and particularly the TAMU extractant, it is likely that they are less soluble in water which decreases their concentration as dissolved P in runoff.

Estimation of DP Using Various STP Methods

The importance of estimating runoff DP concentration as a way of predicting total P losses from a site that had dairy, feedlot manure or effluent is illustrated by Figure 18. Where DP concentrations could reliably ($r^2 = 0.84$) be used to estimate total dissolved P, and to a lesser degree total P losses due to runoff, regardless of management or soil texture properties, both affected infiltration rate. Unfortunately, hopes of using STP to estimate DP concentrations proved unsuccessful. While we did not expect a single highly correlated relationship for all twenty-three sites because of physical differences in soil series and management practices (vegetative cover mostly), our results showed no significant correlation between DP and any of the extraction methods or depths (Table 27). The relationship of STP to actual DP losses (data not shown) showed even less of a correlation and was not statistically significant for any method or depth. This is not surprising since DP loads are dependent not only on DP concentration but also on runoff volume which is variable even from plot to plot within a site.

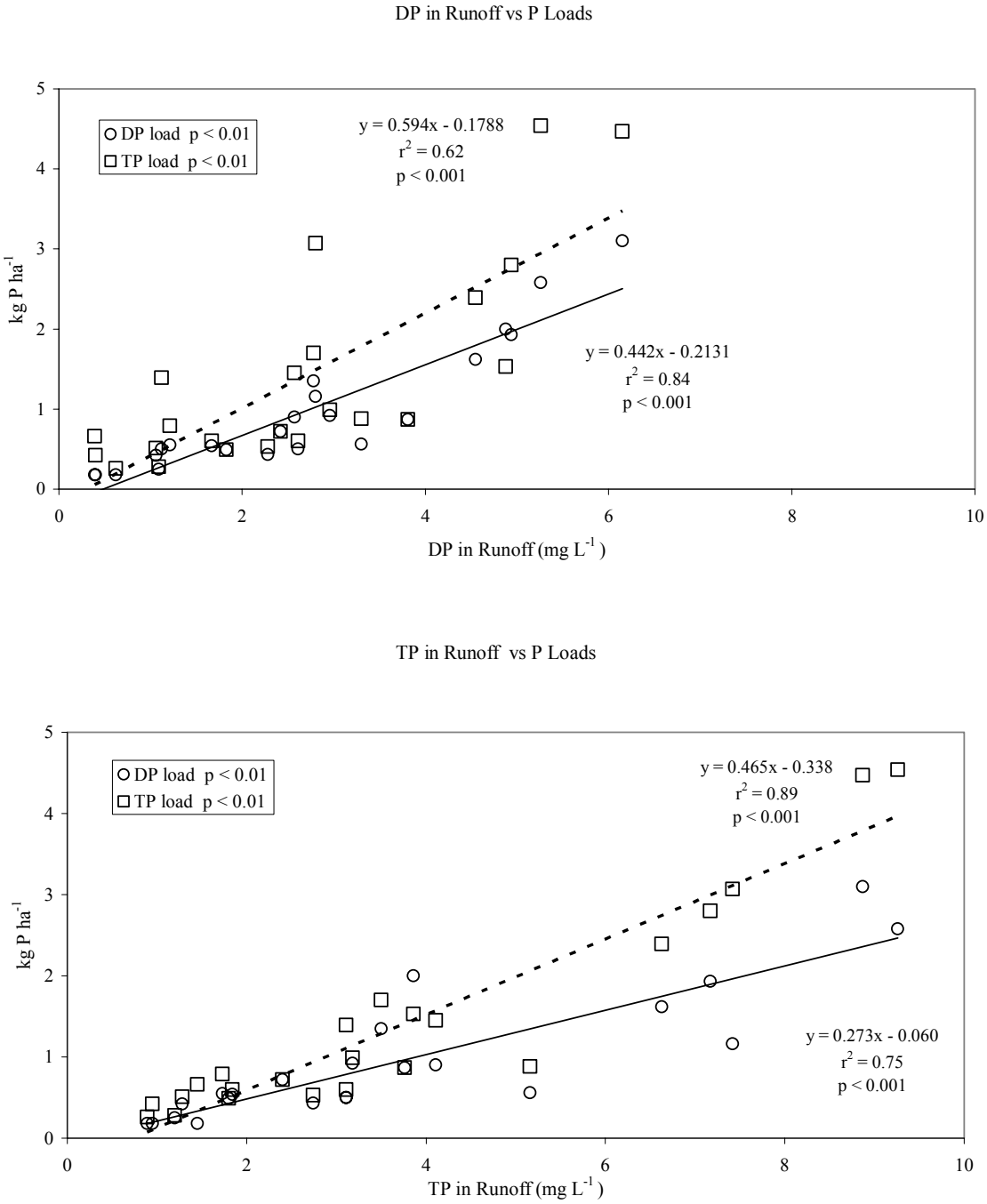

Fig. 18. Relationship between dissolved (DP) and total P (TP) concentrations in runoff and DP and TP loads (losses) for 23 cattle CAFOs across Texas.

Table 27. Linear regression, level of significance, and coefficient of determination for soil test P (mg kg^{-1}) relationships with runoff dissolved P (mg L^{-1}) .

				Soil pH 4.9-8.6					
	CSSP†	r^2	p	Mehlich III	r^2	p	TAMU‡	r^2	p
0-5 cm	-0.0166x + 2.90	0.018	0.54	0.0008x + 2.26	0.03	0.45	0.0005x + 2.32	0.03	0.41
5-15 cm	0.0063x + 2.59	0.001	0.92	0.0025x + 2.03	0.08	0.20	0.0025x + 1.88	0.14	0.08
0-15 cm	-0.0103x + 2.74	0.002	0.83	0.0021x + 1.99	0.08	0.20	0.0015x + 2.03	0.10	0.14
				Soil pH 4.9 - 6.4					
	CSSP	r^2	p	Mehlich III	r^2	p	TAMU	r^2	p
0-5 cm	0.0251x+.69	0.79	0.11	0.0036x+.43	0.90	0.05	0.0023x+.69	0.76	0.13
5-15 cm	0.064x+.63	0.91	0.04	0.0052x+.62	0.82	0.10	0.0039x+.77	0.71	0.16
0-15 cm	0.0478x+.65	0.87	0.07	0.0044x+.58	0.84	0.09	0.0031x+.75	0.72	0.15
				Soil pH 6.8 - 7.5					
	CSSP	r^2	p	Mehlich III	r^2	p	TAMU	r^2	p
0-5 cm	0.0146x+1.98	0.004	0.83	0.0015x+1.357	0.16	0.18	0.0006x+1.72	0.10	0.29
5-15 cm	0.0518x+1.82	0.020	0.63	0.002x+1.63	0.10	0.30	0.0014x+1.72	0.10	0.30
0-15 cm	0.0072x+2.13	0.001	0.94	0.0021x+1.45	0.13	0.22	0.0011x+1.71	0.11	0.28
				Soil pH 7.6 - 8.6					
	CSSP	r^2	p	Mehlich III	r^2	p	TAMU	r^2	p
0-5 cm	0.0451x+4.04	0.01	0.86	0.0029x+3.52	0.16	0.44	0.0034x+2.86	0.33	0.24
5-15 cm	0.0928x+3.86	0.03	0.73	0.0045x+3.42	0.26	0.31	0.0042x+2.96	0.34	0.22
0-15 cm	0.1003x+3.70	0.04	0.69	0.0038x+3.42	0.23	0.33	0.0042x+2.76	0.38	0.19
				Windthorst Soil					
	CSSP	r^2	p	Mehlich III	r^2	p	TAMU	r^2	p
0-5 cm	-0.07x + 2.65	0.24	0.34	-0.0004x + 2.01	0.01	0.87	-0.0005x + 2.10	0.04	0.73
5-15 cm	-0.1835x + 2.77	0.35	0.23	-0.0032x + 2.35	0.18	0.42	-0.0032x + 2.34	0.22	0.35
0-15 cm	-0.1221x + 2.68	0.33	0.25	-0.0019x + 2.29	0.09	0.57	-0.0020x + 2.33	0.14	0.48

† CSSP, calcium solution soluble P

‡ TAMU, acidified ammonium acetate-EDTA extracted P

Attempts to group different soils by dividing DP concentrations by runoff volume, which had been used on Ultisols by other groups (Pote et al., 1999) failed to improve this relationships. In a case when STP and DP were plotted using only six sites which had been identified as a Windthorst soil series but ranged in STP, total soil P (69.8 - 706.4 mg kg^{-1}), pH (4.86 - 7.58), and management practices (vegetation, till system) yielded negative relationships for DP as STP values increased for all methods and depths. Most of these regressions were not significant ($p \geq 0.1$) and those that were could explain very little of the variability as reflected by the r^2 in Table 27.

Separating sites into three groups based on soil pH improved the STP to runoff DP relationship for plots under the lowest pH regimen. Regressions in the acidic to neutral pH range (pH 4.9 - 6.4) had the highest level of correlation (Fig. 19), with r^2 ranging from 0.71 to 0.91 for all extractants and depths. Our correlations using Mehlich III STP values and DP concentrations were higher but comparable to those reported by Schroeder et al., (2004) under acid soil conditions (pH 4.8 to 5.7). However in those experiments analyses were restricted to one soil series and sampling depth did not exceed 10 cm. Both our Mehlich III and CSSP regression lines were slightly better correlated than TAMU P. All other trends between DP and STP in the range of 6.8 - 7.5 and 7.6 - 8.6 were either not significant ($p \geq 0.1$) or had extremely high levels of variability (Table 27). Only the 7.6 to 8.6 soil pH range produced a significant regression line, of which 33 to 38% of the variability could be accounted for using TAMU P. The level of variability which tended to increase with soil depth was not consistently present for sites within working CAFOs. Unlike previous experiments,

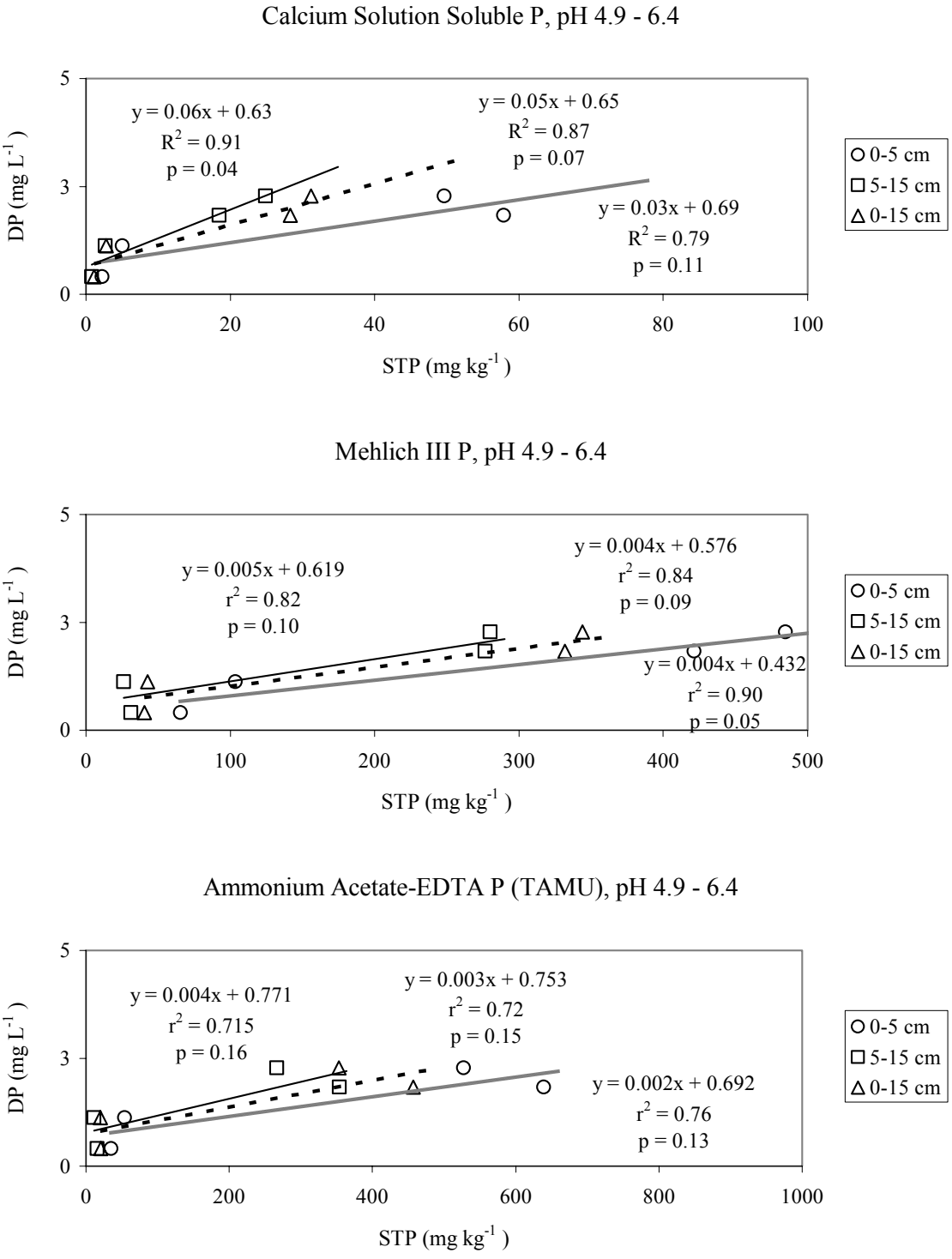

Fig. 19. First-order relationships between runoff dissolved P (DP) and soil test P (STP) from various methods at three soil-sampling depths.

some of the plots had free roaming livestock and in many cases there were obvious disturbances to plots overnight. It is likely that during soil collection some hot spots were inadvertently sampled, which would increase variability in the 0-5-cm cores, while the deeper cores would tend to dilute the effect, thereby masking any differences in variability between the regression lines. There were at least two other factors that could have influenced levels of runoff DP in previous trials. One was that those sites had not had manure applied on a continuous basis as reflected by the background STP level. The second was that an equilibration period of 18-months was used from the time of application to rainfall simulation. Many of the dairies in Texas currently only apply effluent on site and dispose of manure offsite, thereby allowing for a greater distribution throughout the top 15 cm. Current practices call for soil sampling 0 to 5 and 5 to 15 cm for surface applied solids and to 0-15 cm for incorporated solids or surface applied effluents in CAFOs. Our studies showed little improvement in the variability for regression lines obtained using surface STP values, and there were no cases where regression lines for plots that were not significant ($p \leq 0.1$) at the 0-15 cm depth became significant in their prediction of DP when 0-5 cm values were used.

Based on results from the study in Chapter III, we expected DP concentrations to be lower in highly calcareous soils than in soils with equivalent STP but little $CaCO_3$, as precipitation of P by Ca is likely. Although there appeared to be a reduction in amounts of DP as extractable soil Ca concentrations increased, the concentrations of DP were much greater than in previous cases. For example, in our previous findings, a Houston Black series with a STP of 217 g kg^{-1} (TAMU, 0-15 cm) had a DP concentration of 0.41

mg L^{-1}, but a Windthorst series with STP 204 g kg^{-1} had 3.81 mg L^{-1} DP in runoff. Even when extractable Ca for one series was 211 g kg^{-1}, the DP concentration in runoff still averaged 2.4 mg L^{-1}. One possible explanation is that previous experiments used colorimetric techniques that only measure orthophosphate while the use of ICP includes soluble organic P forms. Given that the DP concentrations we obtained tended to be less than 6.1 mg L^{-1}, it is conceivable that soluble organic forms could be interfering with the relationship. Maybe a longer rainfall event could address this problem by diluting the effects of soluble organic forms. Also, the Houston Black series had 18 months between application and simulation, whereas the Windthorst soil had cattle grazing on site during the simulation.

Relationship of DP to Extractable Mg

While previous experiments had a high level of precision in estimating DP in runoff for individual soil series based on STP values, the same trend was not present in sites of working CAFOs. In an attempt to address this problem we tried to find a relationship between other common soil extractable elements that were likely to interact with P. In addition to DP in runoff, soil test P showed no relationship to soluble Ca or Mg (Fig. 20) in runoff. Attempts to relate soil extractable Ca to DP from all sites proved to be poorly correlated with r^2 coefficients of only 0.14 to 0.15 (Fig. 21). However, the relationship was much greater between DP and extractable Mg, with DP increasing proportionally to levels of extractable Mg (Fig. 22). In fact, soil extractable Mg was also highly correlated to Ca. This would indicate that Ca and Mg may be involved in the release of soluble P into runoff. Studies by Booram et al. (1975) and Ferguson et al.

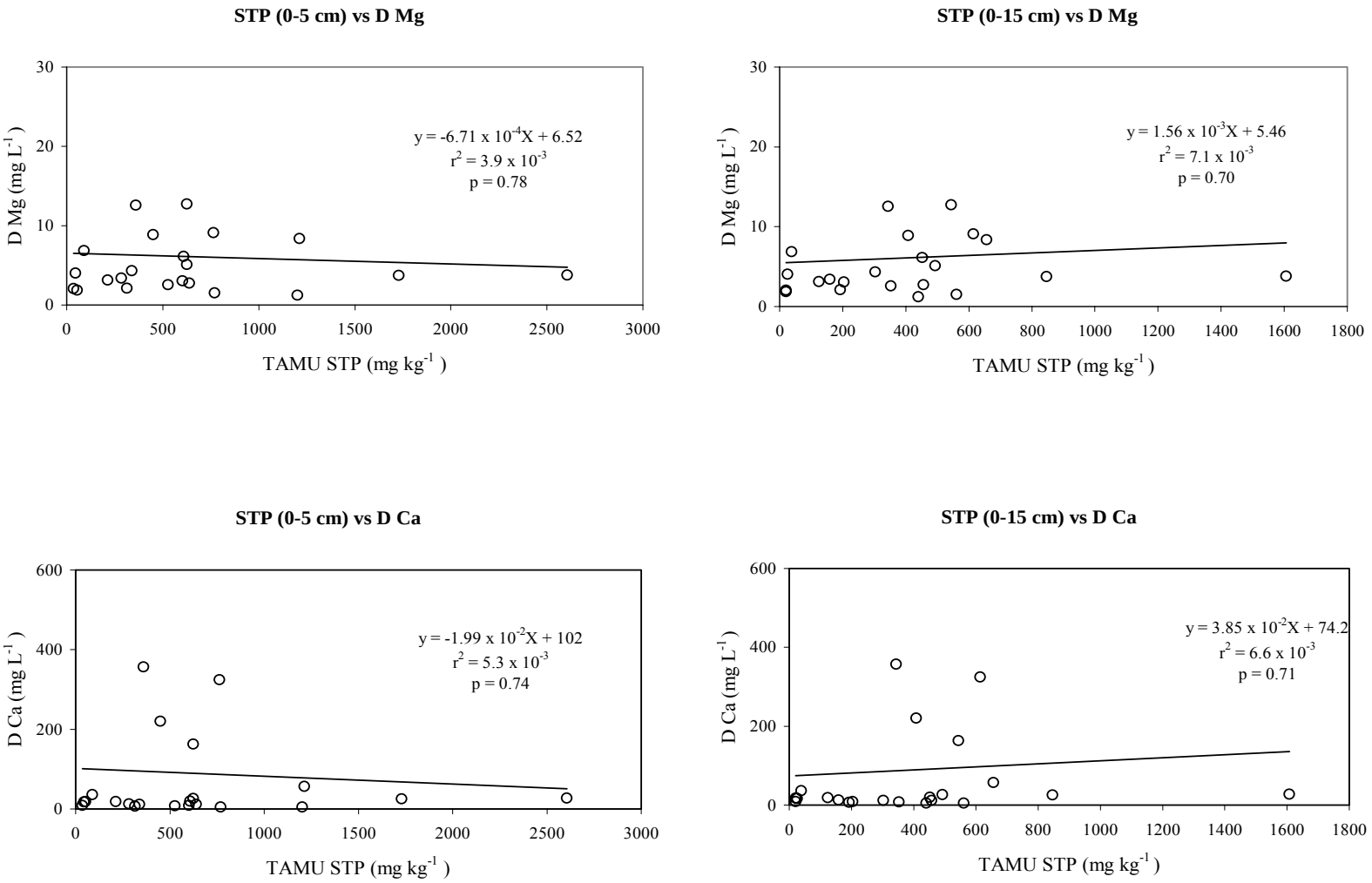

Fig. 20. Relationship between TAMU extractable soil test P (STP) and dissolved (D) Mg and Ca in runoff.

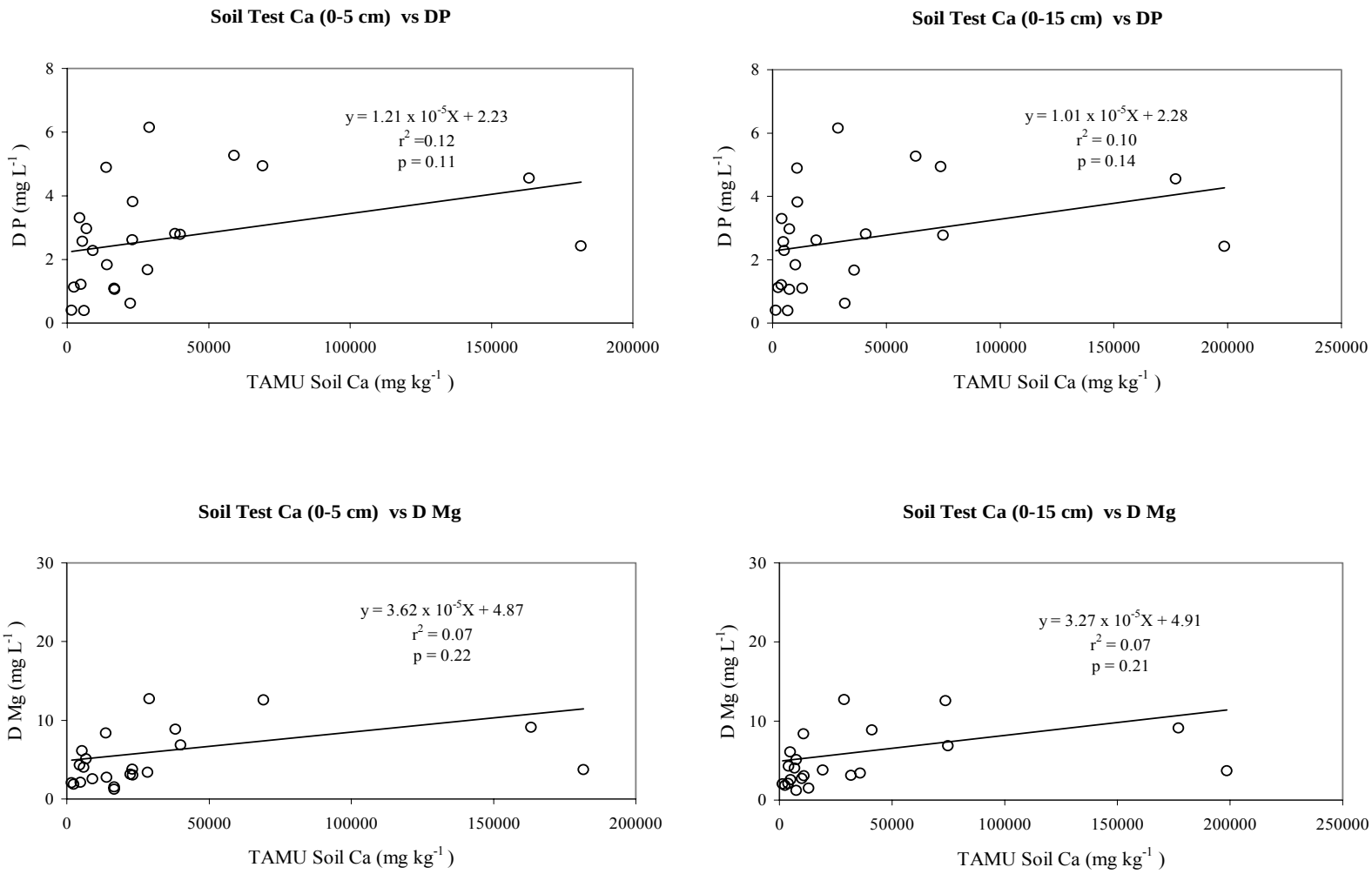

Fig. 21. Relationship between TAMU extractable soil Ca and dissolved (D) Mg and P in runoff.

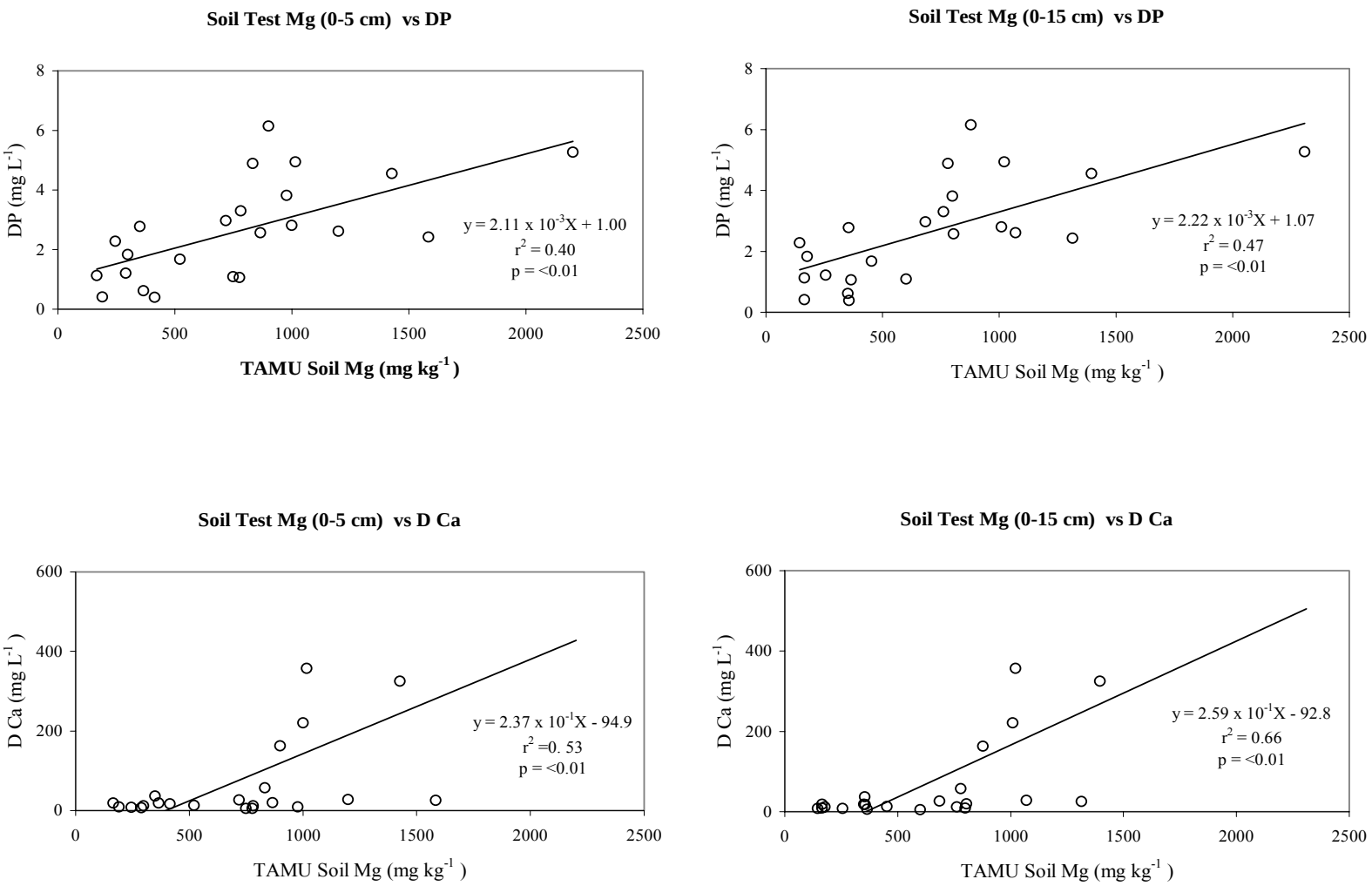

Fig. 22. Relationship between TAMU extractable soil Mg and dissolved (D) P and Ca in runoff.

(1973) indicated that when manure in lagoons reaches equilibrium, several stable Ca^{2+} and Mg^{2+} P compounds are formed, with $MgNH_4PO_4 \cdot 6H_2O$ possibly being one of the major forms. It is also possible that other forms bound to Ca^{2+} and Mg^{2+} are precipitating. This relationship is probably not restricted to manure collected from lagoons, since recent comparisons using nuclear magnetic resonance spectroscopy and conventional fractionation found no differences for P forms found in manure found in lagoons or stockpiled (Hansen et al, 2004). This is of particular importance because Mg-bound phosphates are more soluble than calcium phosphates. The application of manure to soils can also alter mineralization rates through the addition of large amounts of Mg, since it is well established that Mg increases the activity of every kinase enzyme (Jackson et al., 1967). This in turn could be promoting microbial activity and increasing the release of DP into the runoff. If this relationship can be verified, extractable Mg could prove more helpful in estimating losses of P across different soil series and management practices. Not only was this relationship present with respect to runoff concentrations of both DP and TP but also with total losses from these fields.

Comparison Between Colorimetric and ICP Soil Test P

Given that most animal manures have an average C:P ratio of about 8:1 (Sharpley and Moyer, 2000), and that mineralization of P is prevalent when the C:P ratio of materials in soils is below 200 (Pierzynski et al., 2000; Sharpley and Smith, 1989), we would expect the transformation of organic bound P and eventual release the inorganic form. The result would be such that, given enough time, there should be little difference between STP values determined by colorimetric or ICP techniques. Mehlich III P

concentrations for 0-15 cm soil samples using ICP were greater in 20 of 23 sites (Table 28). Readings by ICP could be as much as 67% higher, which would seem to indicate that a significant portion of soil extractable P remains in a soluble organic form since the time between application of manure and rainfall simulation for these sites ranged from a couple of months to over five years. Previous assessments of the organic P fraction of dairy manure have shown that it can be as much as 30% of the total P content (Hansen et al., 2004) and as much 23% of the water extractable P portion (He et al., 2004). This agrees with findings by Labhsetwar and Soltanpur (1985) where ICP and colorimetric readings were compared for Olsen ($NaHCO_3$), $CaCl_2$ extractable, and Na_2-EDTA extractable P, although concentrations were quite low, with the highest value being 35 mg P kg^{-1}. On the other hand, differences between colorimetric and ICP analysis for TAMU solutions were dependent on extractable P concentrations. Phosphorus ICP readings were lower than colorimetric analyses when extractable concentrations were below 350 mg P kg^{-1} soil then increased with increasing concentrations to give readings as much as 53% greater than those obtained using colorimetric measurements (Table 29). One explanation for this behavior could be that an inadvertently chemical reaction between compounds from the TAMU solution and the molybdenum, thereby giving higher estimates of P at lower concentrations for colorimetric reading. However, as P concentrations increased, larger amounts of soluble organic P were probably analyzed by ICP, thereby dwarfing the effect of nonspecific bindings of molybdenum in colorimetric analysis.

Table 28. Mehlich III soil test P analyzed by ICP and colorimetric methods (average of triplicates).

		Colorimetric-P			ICP-P			
Plot	Soil pH	average mg kg^{-1}	sd	cv	average mg kg^{-1}	sd	cv	% change
V	8.57	299.2	9.0	3.0	257.1	27.1	10.6	-16.4
H	7.03	253.4	59.9	23.6	232.5	73.5	31.6	-9.0
U	5.52	40.6	1.8	4.4	40.5	1.1	2.6	-0.2
A	7.84	179.1	14.3	8.0	185.0	35.3	19.1	3.2
J	6.78	38.7	0.4	1.1	44.7	13.0	29.0	13.5
K	7.58	38.5	0.4	1.1	46.1	5.2	11.3	16.6
Q	4.86	31.4	1.4	4.6	42.6	6.9	16.2	26.3
W	8.10	316.0	8.2	2.6	329.2	33.8	10.3	4.0
N	7.51	60.6	2.6	4.2	83.6	3.3	3.9	27.5
T	6.83	327.7	100.5	30.7	350.8	115.1	32.8	6.6
M	7.51	36.9	0.2	0.4	64.5	1.1	1.7	42.8
B	7.92	162.6	3.6	2.2	193.3	55.5	28.7	15.9
S	6.87	158.0	0.3	0.2	199.6	67.7	33.9	20.8
O	6.36	277.8	44.5	16.0	344.0	81.1	23.6	19.3
P	6.33	255.3	10.8	4.2	331.9	103.3	31.1	23.1
R	7.12	396.5	60.7	15.3	521.2	231.4	44.4	23.9
C	7.34	171.3	0.8	0.5	299.1	35.0	11.7	42.7
I	7.10	472.3	67.5	14.3	656.2	28.3	4.3	28.0
F	6.92	212.5	1.6	0.7	397.4	15.2	3.8	46.5
L	7.80	367.7	9.8	2.7	562.0	106.7	19.0	34.6
E	7.26	189.3	2.7	1.4	426.8	39.2	9.2	55.6
G	6.97	444.4	142.3	32.0	771.8	349.3	45.3	42.4
D	7.03	206.4	1.5	0.7	615.2	194.6	31.6	66.5

The relationship between colorimetric and ICP determined STP for both Mehlich III and TAMU extraction were highly correlated under linear regressions with r^2 of 0.76 and 0.83, respectively (Fig. 23). Phosphorus ICP readings were lower than colorimetric analyses when extractable concentrations were below 350 mg P kg^{-1} soil then increased with increasing concentrations to give readings as much as 53% greater than those obtained using colorimetric measurements (Table 29). One explanation for this behavior could be that an inadvertently chemical reaction between compounds from the TAMU solution and the molybdenum, thereby giving higher estimates of P at lower concentrations for colorimetric reading. However, as P concentrations increased, larger amounts of soluble organic P were probably analyzed by ICP, thereby dwarfing the effect of nonspecific bindings of molybdenum in colorimetric analysis.

The relationship between colorimetric and ICP determined STP for both Mehlich III and TAMU extraction were highly correlated under linear regressions with r^2 of 0.76 and 0.83, respectively (Fig. 23).

Table 29. TAMU soil test P analyzed by ICP and colorimetric methods (average of triplicates).

		Colorimetric-P			ICP-P			
Plot	Soil pH	average mg kg^{-1}	sd	cv	average mg kg^{-1}	sd	cv	% change
U	5.52	148.6	9.1	6.2	20.9	0.7	3.5	-610.4
N	7.51	265.4	76.2	28.7	159.5	1.2	0.8	-66.4
V	8.57	421.1	3.3	0.8	344.1	5.8	1.7	-22.4
S	6.87	247.8	34.0	13.7	192.9	6.7	3.5	-28.5
Q	4.86	74.6	46.3	62.0	20.1	1.2	6.0	-270.9
H	7.03	249.2	3.1	1.2	204.1	12.6	6.2	-22.1
C	7.34	336.5	41.9	12.5	302.9	0.8	0.3	-11.1
K	7.58	54.7	0.9	1.6	38.1	1.7	4.5	-43.5
J	6.78	41.9	0.4	1.0	26.0	1.1	4.3	-61.1
M	7.51	130.0	39.7	30.5	124.6	1.5	1.2	-4.3
L	7.80	604.4	4.4	0.7	614.3	4.4	0.7	1.6
O	6.36	337.3	81.8	24.2	353.0	0.7	0.2	4.4
A	7.84	385.3	9.1	2.4	408.0	13.8	3.4	5.6
B	7.92	382.0	3.2	0.8	416.9	8.0	1.9	8.4
W	8.10	492.1	1.1	0.2	543.8	3.7	0.7	9.5
R	7.12	497.2	89.3	18.0	560.4	5.4	1.0	11.3
D	7.03	375.5	12.5	3.3	452.6	12.2	2.7	17.0
T	6.83	360.7	79.9	22.1	440.0	11.4	2.6	18.0
E	7.26	405.1	1.7	0.4	492.7	3.8	0.8	17.8
P	6.33	335.2	17.1	5.1	457.0	19.6	4.3	26.6
F	6.92	468.0	6.3	1.3	657.0	16.3	2.5	28.8
I	7.10	638.1	8.4	1.3	846.0	8.8	1.0	24.6
G	6.97	758.0	20.5	2.7	1606.8	74.9	4.7	52.8

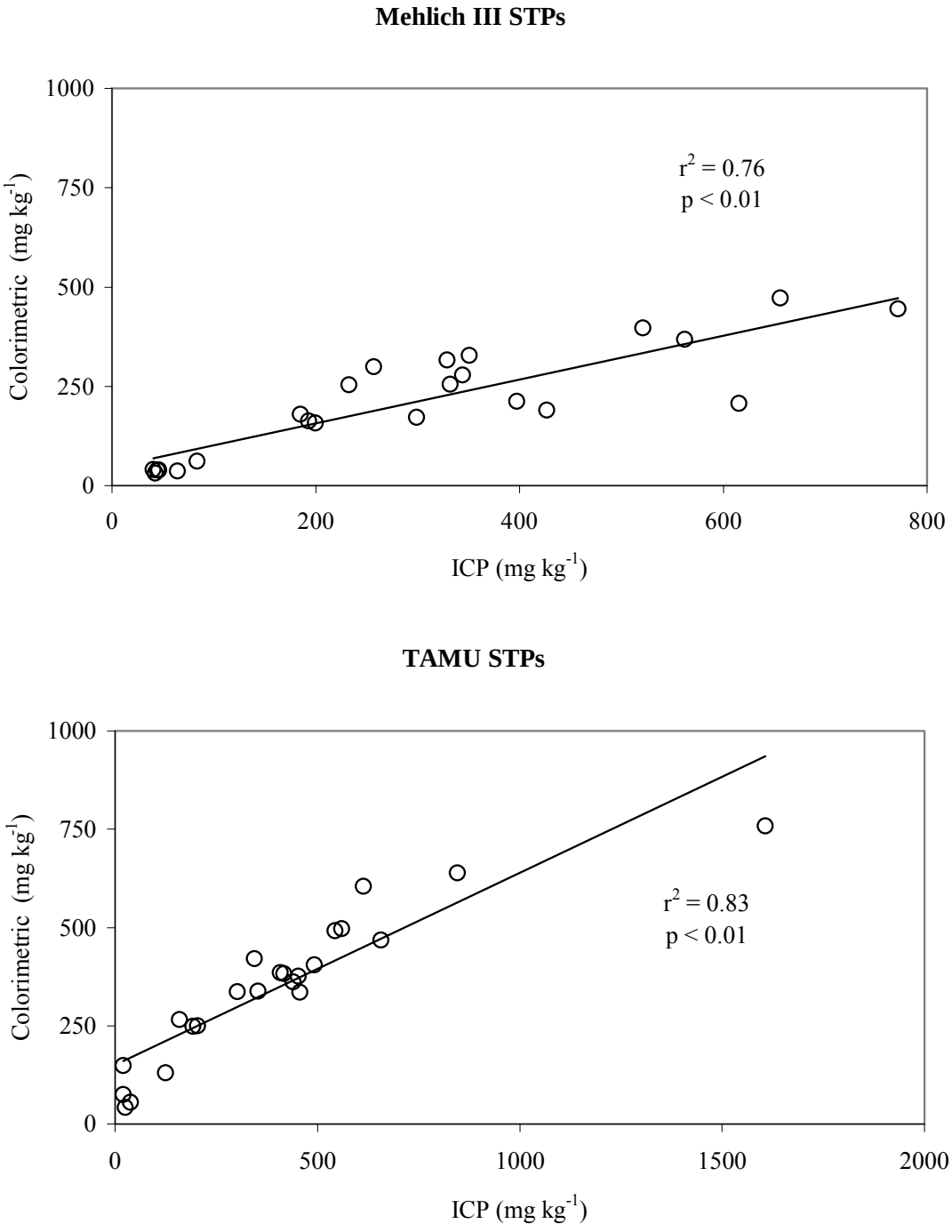

Fig. 23. Correlation of Mehlich III and TAMU extractable P determined colorimetrically and by ICP.

Conclusions

Predicting potential losses from individual fields appears to be primarily influenced by management practices, which have the greatest influence on the runoff and erosion of P. Unlike results of Chapter III, estimation of dissolved P in runoff using STP was not possible for plots in CAFO fields, except when soil pH conditions were acid to slightly neutral (pH 4.8-6.4). The relationship appeared to be more dependent on soil pH range than on soil characteristics. No consistent difference in variability was obtained whether soil sampling was restricted to 0-5 or 0-15 cm. The TAMU P extraction method gave the highest correlation to DP in soils of low and high (calcareous) pH values. It appeared that management, total soil P content, the period between manure application and simulation were the primary factors affecting the release of DP into runoff. Part of the lack of a relationship might be due to the sensitivity of contributions to DP from soluble organic P that vary depending on the incubation time between application and runoff. The ability of small amounts of soluble organic P to alter the linear relationship can be illustrated with Mehlich III P which ranged from 65 to 1221 mg kg^{-1} (0-5 cm depth) while the lowest and highest DP concentrations in runoff were 0.4 and 6.1 mg L^{-1}. Estimation of DP from all sites, regardless of pH, total soil P, STP values, or management practice, was better correlated to extractable soil Mg. This could indicate that a major portion of the soluble P found in runoff is actually dependent on Mg concentrations.

CHAPTER V

CORRELATION OF P LOSSES TO TEXAS P INDEX RATING

Synopsis

Current state regulations call for the development of nutrient management plans for all CAFOs before land application of manure. One major component of these plans involves the Phosphorus Index (PI), an assessment tool used to rate sites where manure is to be applied based on the likelihood of the soils to contribute P to streams and lakes due to runoff losses. This chapter's main goal was to verify if the current parameters used in PI ratings reflect actual P losses as measured in the previous experiments (Chapter IV) from cattle CAFOs with different characteristics due to management, location, and soil type.

Materials and Methods

Phosphorus Index Rating

The current Texas PI rating system is outlined in Tables 30 and 31, for West and East Texas, respectively. Final ratings were obtained by multiplying each characteristic value by its weighting factor and then summing. Soil test ratings using two sampling depths (0-5 and 0-15 cm) were based on crop response recommendations by the TCE Soil, Water, and Forage Testing Laboratory for Mehlich III and TAMU extractions. CAFO producers provided information regarding manure/effluent application rates and

Table 30. Phosphorus index worksheet for West Texas (USDA/NRCS, 2000b).

PHOSPHORUS INDEX WORKSHEET FOR WEST TEXAS

PHOSPHORUS INDEX WORKSHEET for West Texas						
Client Name:			Field(s):		Date:	
Planner:			Location:		Crop:	
Impaired Watershed (Y or N):			Runoff Curve No.:		% Slope:	
Site Characteristic (Weighting Factor)	[Weighting Factor Times the Column Factor] 0	1	2	4	8	Sub Total
Soil Test P Rating	N/A	Very Low – Low	Moderate	High	Very High	
(1.00)	[0]	[1.0]	[2.0]	[4.0]	[8.0]	
Fertilizer Phosphorus (P_2O_5) Application Rate	None Applied	1-40 lbs/ac P_2O_5	41-90 lbs/ac P_2O_5	91-150 lbs/ac P_2O_5	>150 lbs/ac P_2O_5	
(0.75)	[0]	[0.75]	[1.5]	[3.0]	[6.0]	
Organic Phosphorus (P_2O_5) Application Rate	None Applied	1-40 lbs/ac P_2O_5	41-90 lbs/ac P_2O_5	91-150 lbs/ac P_2O_5	>150 lbs/ac P_2O_5	
(0.75)	[0]	[0.75]	[1.5]	[3.0]	[6.0]	
Phosphorus Fertilizer Application Method and Timing	None Applied	Placed deeper than 2 in. or broadcast and incorporated within 48 hours	Incorporated immediately before planting	Incorporated >4 months before planting, or surface applied < 4 months before planting	Surface applied >4 months before planting	
(0.50)	[0]	[0.50]	[1.0]	[2.0]	[4.0]	
Organic Phosphorus source Application Method and Timing	None Applied	Placed deeper than 2 in. or broadcast and incorporated within 48 hours	Incorporated immediately before planting	Incorporated >4 months before planting, or surface applied < 4 months before planting	Surface applied >4 months before planting	
(0.50)	[0]	[0.50]	[1.0]	[2.0]	[4.0]	
Proximity of nearest field edge to named stream or lake	> 2000 feet	1000 – 1999 feet	500 – 999 feet	100 – 499 feet	< 100 feet	
(1.25)	[0]	[1.25]	[2.5]	[5.0]	[10.0]	
Runoff Class (Runoff Class Table 3)	Negligible	Low	Moderate	High	Very High	
(1.00)	[0]	[1.0]	[2.0]	[4.0]	[8.0]	
Soil Erosion (all sources)	Very Low <1 t/ac	Low 1-3 t/ac	Medium 3-5 t/ac	High 5-10 t/ac	Very High >10 t/ac	
(1.50)	[0]	[1.5]	[3.0]	[6.0]	[12.0]	
					Total Index Points:	

Phosphorus Index Classification – West Texas			
Index Pts.	P Runoff Potential	Non-impaired Critical P Level (ppm)	Impaired Critical P Level (ppm)
< 15	Very Low - Low	500	300
15 – 24.75	Medium	400	250
25 - 35	High	300	200
> 35	Very High	200	200

Table 31. Phosphorus index worksheet for East Texas (USDA/NRCS, 2000b).

PHOSPHORUS INDEX WORKSHEET FOR EAST TEXAS

PHOSPHORUS INDEX WORKSHEET for East Texas						
Client Name:		Field(s):		Date:		
Planner:		Location:		Crop:		
Impaired Watershed (Y or N):		Runoff Curve No.:		% Slope:		
Site Characteristic (Weighting Factor)	[Weighting Factor Times the Column Factor] 0	1	2	4	8	Sub Total
Soil Test P Rating	N/A	Very Low – Low	Moderate	High	Very High	
(1.00)	[0]	[1.0]	[2.0]	[4.0]	[8.0]	
Fertilizer Phosphorus (P_2O_5) Application Rate	None Applied	1-40 lbs/ac P_2O_5	41-90 lbs/ac P_2O_5	91-150 lbs/ac P_2O_5	>150 lbs/ac P_2O_5	
(0.75)	[0]	[0.75]	[1.5]	[3.0]	[6.0]	
Organic Phosphorus (P_2O_5) Application Rate	None Applied	1-40 lbs/ac P_2O_5	41-90 lbs/ac P_2O_5	91-150 lbs/ac P_2O_5	>150 lbs/ac P_2O_5	
(0.75)	[0]	[0.75]	[1.5]	[3.0]	[6.0]	
Phosphorus Fertilizer Application Method and Timing	None Applied	Placed deeper than 2 in. or broadcast and incorporated within 48 hours	Surface applied 12/1-2/15	Surface applied 2/16-4/15 or 6/16-11/30	Surface Applied 4/16-6/15	
(0.50)	[0]	[0.50]	[1.0]	[2.0]	[4.0]	
Organic Phosphorus source Application Method and Timing	None Applied	Placed deeper than 2 in. or broadcast and incorporated within 48 hours	Surface applied 12/1-2/15	Surface applied 2/16-4/15 or 6/16-11/30	Surface Applied 4/16-6/15	
(0.50)	[0]	[0.50]	[1.0]	[2.0]	[4.0]	
Proximity of nearest field edge to named stream or lake	> 2000 feet	1000 – 1999 feet	500 – 999 feet	100 – 499 feet	< 100 feet	
(1.25)	[0]	[1.25]	[2.5]	[5.0]	[10.0]	
Runoff Class (Runoff Class Table 3)	Negligible	Low	Moderate	High	Very High	
(1.00)	[0]	[1.0]	[2.0]	[4.0]	[8.0]	
Soil Erosion (all sources)	Very Low <1 t/ac	Low 1-3 t/ac	Medium 3-5 t/ac	High 5-10 t/ac	Very High >10 t/ac	
(1.50)	[0]	[1.5]	[3.0]	[6.0]	[12.0]	
					Total Index Points:	

Phosphorus Index Classification – East Texas			
Index Pts.	P Runoff Potential	Non-impaired Critical P Level (ppm)	Impaired Critical P Level (ppm)
< 12	Very Low - Low	500	300
12 – 22.75	Medium	400	250
23 - 32	High	300	200
> 32	Very High	200	200

methods on a mail-in survey. When analysis of manure or effluent was not provided, TCE values for P concentrations were used (Feagley et al., 2004). Proximity to surface water was measured as distance from to the nearest named body of water to the location where manure or effluent application ceased using the most recent soil survey maps from the appropriate counties. Runoff class evaluation included measured slope, vegetative cover evaluation conducted on site, and soil hydrological class as provided in the soil survey. Yearly soil erosion estimates were obtained through the use of the NRCS Revised Universal Soil Loss Equation, Version 2 (RUSLE2, 2004) software for water erosion only. We also calculated total sediment eroded and suspended materials in runoff (sediment and organic matter) lost from fields by multiplying plot losses for a single event times the number of events that would be expected for that specific county (average yearly rainfall/total applied rainfall for one rainfall event). Relationships were evaluated by plotting PI ratings for each site (average of three replicates) against total measured P and dissolved P losses from simulated rainfall trials.

Results and Discussion

Management Effects on Sediment and P Losses

There was a wide range in measured losses of eroded sediment and suspended material (0- 57.2 Mg ha^{-1} yr^{-1}), DP (3.0- 46.8 kg P ha^{-1}) and TP (4.4- 67.5 kg P ha^{-1}) from twenty-three sites on the Texas Southern High Plains and the Central Texas region

Table 32. Sediment erosion rates, dissolved P, and total P losses from CAFOs across Texas.

Plot	Vegetative cover/ management	Measured sediment (Mg ha^{-1} yr^{-1})	RUSLE2 estimate (Mg ha^{-1} yr^{-1})	DP load (kg P ha^{-1} yr^{-1})	TP load (kg P ha^{-1} yr^{-1})
A	Crop residue	7.95	1.14	12.01	31.79
B	Crop residue	43.61	4.91	28.76	50.61
C	Grass	0.68	0.22	5.49	8.59
D	Row-crop	8.47	0.45	9.40	15.04
E	Tilled	13.57	0.36	9.71	10.45
F	Row-crop	30.55	21.21	38.38†	29.43†
G	Row-crop	5.58	4.01	9.21	11.18
H	Grass	0.14	0.21	11.94	11.91
I	Grass	3.80	0.31	10.15	10.13
J	Row-crop	12.31	51.52	3.32	12.37
K	Grass	6.16	0.18	24.99	31.48
L	Crop residue	16.65	7.17	29.50	43.63
M	Grass	5.79	0.19	3.09	4.44
N	Grass	3.06	0.17	8.82	9.70
O	Grass	0.00	0.25	6.58	8.05
P	Grass	0.00	0.34	8.02	8.00
Q	Crop residue	50.75	3.67	9.77	27.10
R	Grass	0.48	0.21	4.33	4.82
S	Crop residue	1.07	2.60	9.66	13.82
T	Grass	0.28	0.15	6.47	7.85
U	Crop residue	11.23	26.88	3.18	7.44
V	Tilled	57.15	45.47	36.27	52.44
W	Tilled	28.01	25.76	46.86	67.53

RUSLE2- Revised universal soil loss equation version 2.0
DP: TP - Dissolved and total P
† Inherent limitations in the total digestion procedure produced similar concentrations for DP and TP (4.88 and 3.86 mg L^{-1}) that appeared larger when scaled to hectare yr^{-1}

(Table 32). While total sediment loss was measured, it was not included as total P loss because it would result in a gross overestimation under most conditions because these materials are not likely to be carried off-site unless flooding occurs. Measured sediment and suspended material lost and the estimates obtained using the RUSLE2 program differed. The difference was greatest for the Southern High Plains. Part of the reason can be attributed to the high intensity of applied rainfall, which for comparison reasons had to be constant for the entire study. Other noticeable differences occurred in grass covered sites and sites with row crop residues. In our experiments, we obtained significantly higher erosion rates than predicted values (average of 2.0 vs. 0.2 Mg ha^{-1} yr^{-1}), but this could be due to the difficulty of separating sediment from the large amounts of organic debris (seeds, and recent depositions of manure from livestock) that were collected during the rainfall events. Additional sediment also could have been contributed from disturbance induced by the process of setting down frames and filling any gaps with loose soil. The underestimation of sediment loss from sites that contain crop residues was primarily due to the fact that values of RUSLE2 are averaged over the course of a year, whereas our measured values were all obtained from rainfall events that had less than 10% vegetative cover (Fig. 24).

The extremely high losses of DP ranging from 3.09 to 46.86 kg P ha^{-1} yr^{-1}, are based on a series of worst case scenarios, where the entire yearly rainfall occurs in events that all have the same high intensity and generate 30-minute runoff volumes. In addition, these estimates do not include changing vegetative cover that occur during the growing season, nor the fact that P concentrations fall drastically with successive rainfall

Fig. 24. Site with crop residue during rainfall simulation.

events. It is not our intent to predict annual P discharge from fields but rather to estimate the greatest losses that could take place. Current studies on swine amended fields by other groups (Daverede et al., 2004) using the same rainfall intensity and simulator have obtained DP and TP loads of 1.3 and 1.7 kg ha^{-1}, respectively, for a single event, while at the other extreme poultry litter application has yielded DP and TP

load values of 0.5 to 5.5 , 1.3 to 8.5 kg ha^{-1} , respectively (Tarkalson and Mikkelsen, 2004) Our results for a single rainfall event fell in between those previously reported, with DP losses of 0.18 to 3.51 kg P ha^{-1}, and TP losses from 0.26 to 4.54 kg P ha^{-1}. In a watershed scale study conducted over a seven-year period, Udawatta et al. (2004) measured TP losses ranging from 0.29 to 3.59 kg ha^{-1} yr^{-1} but also found that 27% of all losses occurred during five of the 66-runoff events. Since there is no consensus to the most relevant way to assess P losses, we decided to report our findings assuming similar multiple rainfall events on a yearly basis.

As expected, management practices influenced losses in the runoff. When sediment, DP, and TP losses were averaged according to type of vegetation and cultivation practices, a clear trend was observed with values increasing in the order from low to high: grasses, row-crops, row-crop residues, and tilled soils (Table 33). Although clear differences due to management practices were observed between means, only grasses and tilled fields were significantly different at p level of 0.5. As mentioned before, RUSLE2 tended to greatly underestimate losses from sites with crop residues, 7.73 Mg ha^{-1} yr^{-1} compared to our observed values 21.88 Mg ha^{-1} yr^{-1}. The use of RUSLE2 estimates for erosion rates was found to adversely influence the relationship between PI rating and runoff P losses collected at the field scale. This will be addressed in the following section.

Table 33. Grouping of P index rating, sediment, dissolved P (DP), and total P (TP) losses by management practices on CAFOs.

	Grass		Row-crop		Crop residue		Tilled	
	Mean	S.D.	Mean	S.D.	Mean	S.D.	Mean	S.D.
Measured sediment (Mg ha^{-1} yr^{-1})	2.04a†	2.46	14.23b	11.23	21.88bc	20.36	32.91c	22.20
RUSLE2 estimate (Mg ha^{-1} yr^{-1})	0.22a	0.06	19.30ab	23.32	7.73b	9.60	23.86a	22.62
DP load (kg P ha^{-1} yr^{-1})	8.99a	6.22	15.08ab	15.79	15.48a	10.98	30.95b	19.13
TP load (kg P ha^{-1} yr^{-1})	10.50a	7.71	17.00b	8.44	29.06c	16.16	43.47c	29.58
Phosphorus Index Rating	17.48a	6.84	22.56ac	7.75	20.44a	7.12	30.71bc	13.32
Phosphorus Index Rating ‡	18.38a	6.98	23.69ab	7.77	24.56b	8.39	33.71b	8.22

†, Within rows, means followed by the same letter are not significantly different at the 0.10 probability level
‡, PI, phosphorus index rating using measured sediment rates

Phosphorus Index Ranking of Sites

Individual points given to each factor for all sites are presented in Table 34. Two different ratings were calculated for each site using STP values from either Mehlich III or TAMU extraction methods. The result was a difference of 4 points in two of twenty-three sites. Phosphorus index vulnerability ratings ranged from 5.56 to 40.3. In general there was a trend of increasing PI values based on the amount of vegetative cover and management practices. The averaged PI rating for all grass-covered sites was the lowest (17.48), followed by those with crop residue cover (20.06), 10-cm tall row crops (22.56), and sites that have been tilled (30.71). There was no statistically significant ($p \leq 0.1$) linear relationship between the PI ranking and total soil P content or extractable levels of P at any sampling depth regardless of method used. Attempts to show a relationship between actual measured losses for dissolved P or total suspended P in the runoff proved less than successful when all sites (dairies and feedlots) were plotted (Table 35). Only between ten to 13 percent of the sample variability could be explained for either DP or TP losses using either TAMU or Mehlich III STP. Part of this can be attributed to the reliability of data provided. Many of the producers that initially agreed to cooperate in the study are currently embroiled in various legal litigations and appeared reluctant to provide all of the information such as manure content, rates, and dates of application. It is also possible that dietary differences between the beef and dairy CAFOs were influencing our results. An overall improvement in the relationship was obtained by separating sites into feedlots, which were located in West Texas and dairies in East Texas.

Table 34. Phosphorus index rating for selected CAFOs in East and West Texas.

	West Texas Freedlots					East Texas Dairies																
	A	B	C	D	E	F	G	H	I	J	K	L	M	N	O	P	Q	R	S	T	U	V	W
Soil Test P (0-15 cm) Mehlich III	8.0	8.0	8.0	8.0	8.0	8.0	8.0	8.0	8.0	2.0	2.0	8.0	4.0	4.0	8.0	8.0	2.0	8.0	8.0	8.0	2.0	8.0	8.0
Soil Test P (0-15 cm) TAMU	8.0	8.0	8.0	8.0	8.0	8.0	8.0	8.0	8.0	1.0	2.0	8.0	8.0	8.0	8.0	8.0	1.0	8.0	8.0	8.0	1.0	8.0	8.0
Inorganic Fertilizer P Application Rate	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.1	1.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Organic P Application Rate	0.0	0.0	0.0	4.5	4.5	4.5	4.5	0.0	4.5	4.5	0.0	4.5	4.5	4.5	4.5	4.5	4.5	0.6	0.0	0.6	0.0	4.5	4.5
Inorganic Fertilizer P Application Method and Timing	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Organic P Application Method and Timing	0.0	0.0	0.0	1.0	1.0	0.3	0.3	0.0	0.5	0.5	0.0	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.0	0.3	0.0	0.3	0.3
Proximity to Stream	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.6	0.0	0.0	1.6	0.0	12.5	12.5	6.3	3.1	3.1	3.1	6.3	6.3	12.5	1.6	1.6
Runoff Class	4.0	8.0	2.0	2.0	2.0	4.0	2.0	4.0	2.0	8.0	2.0	4.0	0.0	0.0	4.0	4.0	8.0	4.0	2.0	2.0	2.0	8.0	4.0
Soil Erosion (RUSLE2)	0.0	2.3	0.0	0.0	0.0	9.0	2.3	0.0	0.0	18.0	0.0	4.5	0.0	0.0	0.0	0.0	2.3	0.0	2.3	0.0	18.0	18.0	18.0
Total (Mehlich III)	12.0	18.3	10.0	15.5	15.5	25.8	17.0	13.6	15.0	33.0	5.6	21.3	21.3	21.3	25.1	22.0	20.1	15.9	18.5	17.1	34.5	40.3	36.3
Total (TAMU)	12.0	18.3	10.0	15.5	15.5	25.8	17.0	13.6	15.0	32.0	5.6	21.3	25.3	25.3	25.1	22.0	19.1	15.9	18.5	17.1	33.5	40.3	36.3

Table 35. Relationship between P index ratings with Mehlich III or TAMU soil test P (STP) and phosphorus loads using RUSLE2 erosion estimates.

Region	P form	STP used	Regression line	r^2	p
All Texas	DP	TAMU	y = 0.472x + 4.816	0.11	0.14
	TP	TAMU	y = 0.652x + 7.234	0.11	0.14
All Texas	DP	Mehlich III	y = 0.516x + 3.951	0.12	0.10
	TP	Mehlich III	y = 0.739x + 5.512	0.13	0.09
West	DP	TAMU	y = 2.097x - 16.804	0.56	0.14
	TP	TAMU	y = 2.988x - 19.292	0.30	0.34
West	DP	Mehlich III	y = 2.097x - 16.804	0.56	0.14
	TP	Mehlich III	y = 2.988x - 19.292	0.30	0.34
East	DP	TAMU	y = 0.468x + 4.467	0.10	0.23
	TP	TAMU	y = 0.791x + 2.224	0.16	0.12
East	DP	Mehlich III	y = 0.520x + 3.368	0.11	0.17
	TP	Mehlich III	y = 0.890x + 0.120	0.19	0.07

The most pronounced improvement was observed with the West Texas CAFOs, which aside from providing a more thorough description of application dates, also had the least variation in soil pH. For East Texas sites, we could explain only 9-11% of DP variability and 15 to 19% of TP losses using our field rankings. We suspected that our experimental procedures could be influencing our lack of fit of the data. Sharpley (1995b) obtained high correlations (r^2=.70) between PI ranking and measured total P losses in 30 watersheds. However, besides differences in scale and monitoring time (16 years), their ranking did not include proximity to streams, runoff class, and their sediment loads were obtained by measuring annual sediment discharges collected in flumes. By substituting estimated RUSLE2 erosion with our measured sediment and suspended material losses per plot, we were able to greatly increase our correlations. Table 36 shows a significant increase in correlation regardless of grouping (feedlot and dairies) or STP method used in the relationship between the P index rating and measured losses of DP and TP. Grouping of West Texas sites (feedlots) had r^2 values of 0.76 and 0.48 for DP and TP loads in runoff, respectively. The improvement was not as drastic for those in the East, with r^2 values of 0.28- 0.30 and 0.34- 0.38 for DP and TP loads in runoff, respectively. Part of the difference for East Texas may be attributed to the much wider range in soil pH values, which would influence elements that bind P. However a more important factor could also be due to the contribution of fresh manure from animals grazing four pastures that we were unable to fence out. Removal of these sites from the East Texas regression netted r^2 values of 0.46 and 0.56 for the DP and TP relationships, respectively.

Table 36. Relationship between P index ratings with Mehlich III or TAMU soil test P (STP) and phosphorus loads using measured erosion rates.

Region	P form	STP used	Regression line	r^2	p
All Texas	DP	TAMU	y = 0.828x - 4.368	0.32	0.005
	TP	TAMU	y = 1.193x - 6.560	0.34	0.003
All Texas	DP	Mehlich III	y = 0.864x - 4.998	0.35	0.003
	TP	Mehlich III	y = 1.265x - 7.934	0.38	0.002
West	DP	TAMU	y =0 .878x - 5.367	0.76	0.055
	TP	TAMU	y = 1.366x - 5.393	0.48	0.197
West	DP	Mehlich III	y =0 .878x - 5.367	0.76	0.060
	TP	Mehlich III	y = 1.366x - 5.393	0.48	0.200
East	DP	TAMU	y = 0.816x - 4.110	0.28	0.025
	TP	TAMU	y = 1.195x - 7.941	0.34	0.012
East	DP	Mehlich III	y = 0.860x - 4.885	0.30	0.020
	TP	Mehlich III	y = 1.278x - 9.543	0.38	0.060

Modification of the P Index

The use of other states' P indices to rank our sites failed to provide a better model for the relationship to our measured P losses. This was partly due to the inclusion of categories such as manure availability, sub-surface drainage, and buffer strips which were not measured during our simulations and therefore were entered as high values in the rating. In an attempt to improve the correlation, a new site characteristic was added in the form of extractable soil Mg. As was previously shown in Chapter IV, there is a clear relationship between extractable soil Mg by the TAMU method and DP in runoff. This relationship while not extremely high was present across all soil pHs and total soil P contents. The weighting factor (1.5) and column rankings used in the STP category were maintained but the concentrations of extractable Mg were not based on crop nutrient requirements. Using the vulnerability class rating points (Table 30 and 31) and fitting them into the linear equations obtained for each region (Table 35 and 36), we estimated expected DP and TP loads. These values are listed in Table 37. Estimated loads were then plotted against extracted soil Mg using a first order regression to obtain the following levels that were used to rank Mg concentration from NA to very high (Table 38). The obtained values were then added to the previous calculated PI points. Since extractable Mg was determined only for the TAMU extraction, PI points use only TAMU STP values.

Table 37. Estimated P losses expected from sites ranked by vulnerability using the phosphorus index classification.

	West Texas			East Texas	
Site Ranking	DP†	TP‡		DP†	TP‡
	—	—	kg P ha^{-1} yr^{-1}	—	—
Very low	< 7.8	< 15.1		< 5.7	< 6.4
Medium	7.8 - 16.4	15.1 - 28.4		5.7 - 14.5	6.4 - 19.2
High	16.6 - 25.4	28.8 - 42.4		14.5 - 14.7	19.2 - 19.5
Very High	> 25.4	> 42.4		> 22.0	> 30.3

† DP; dissolved P.
‡ TP, total P.

Our estimated TP loads were much greater than those reported by Sharpley (1995b). The aforementioned study had low, medium, and high rated classes with total P losses of less than 1, 1-4, and greater than 4 kg P ha^{-1} yr^{-1}, respectively. Of course these values were obtained under natural rainfall for a sixteen-year period at the watershed scale. Because of our design, we are more susceptible to effects such as hotspots, in addition to creating conditions for a worst-case scenario with all storm events having 30-minute runoff periods and high intensity. Additionally, recent studies comparing rainfall intensity on plot of different sizes showed an increase in runoff DP concentration inversely related to plot area. (Sharpley and Kleinman, 2003).

Table 38. Proposed levels of extractable soil Mg.

West Texas — TP Losses

	NA†	Low	Moderate	High	Very High
Soil Mg (mg/kg)	0	<779	779 - 1340	1341 - 1898	>1898
column factor	0.0	1.0	2.0	4.0	8.0

DP Losses

	NA†	Low	Moderate	High	Very High
Soil Mg (mg/kg)	0	<710	711 - 1380	1381 - 2053	>2053
column factor	0.0	1.0	2.0	4.0	8.0

East Texas — DP Losses

Soil Mg (mg/kg)	NA†	Low	Moderate	High	Very High
column factor	0	<54	55 - 567	568 - 996	>996
	0.0	1.0	2.0	4.0	8.0

TP Losses

Soil Mg (mg/kg)	NA†	Low	Moderate	High	Very High
column factor	0	<548	548 - 562	563 - 1100	>1100
	0.0	1.0	2.0	4.0	8.0

† NA, none applied.

Incorporating extractable Mg into our rating system allowed us to strengthen the linear relationship in both West and East Texas regions (Fig. 25). Although the relationship for East Texas were still low (r^2 =0.41 and 0.37), it could be increased to similar levels found in West Texas feedlots if we omitted sites that had cattle grazing in the field during our simulations (Fig. 26). We felt justified in doing this since the frames attracted the herd overnight and influenced the levels of collected P in runoff.

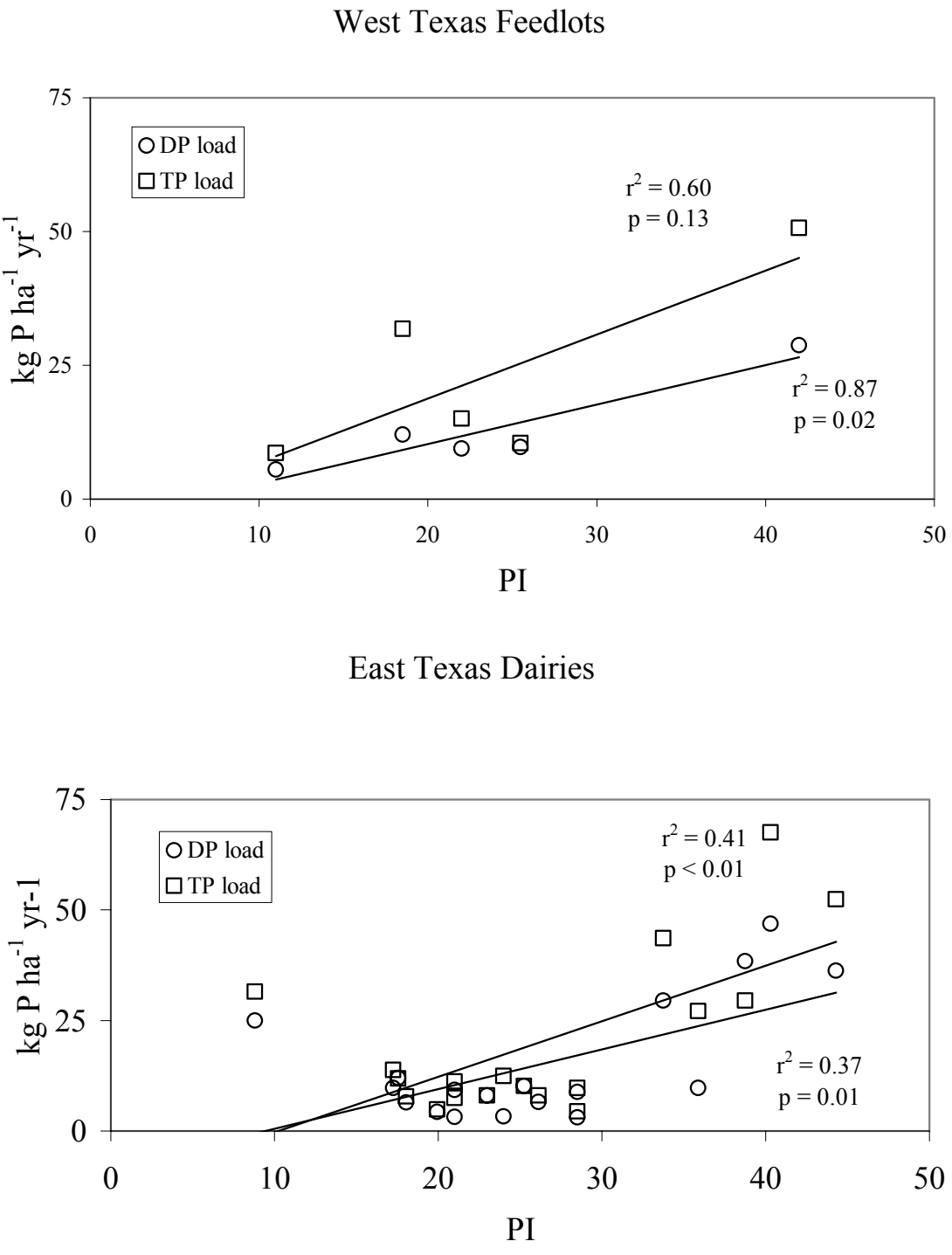

Fig. 25. Relationship between P index (PI) rating (using extractable Mg) and dissolved (DP) and total P (TP) loads in runoff.

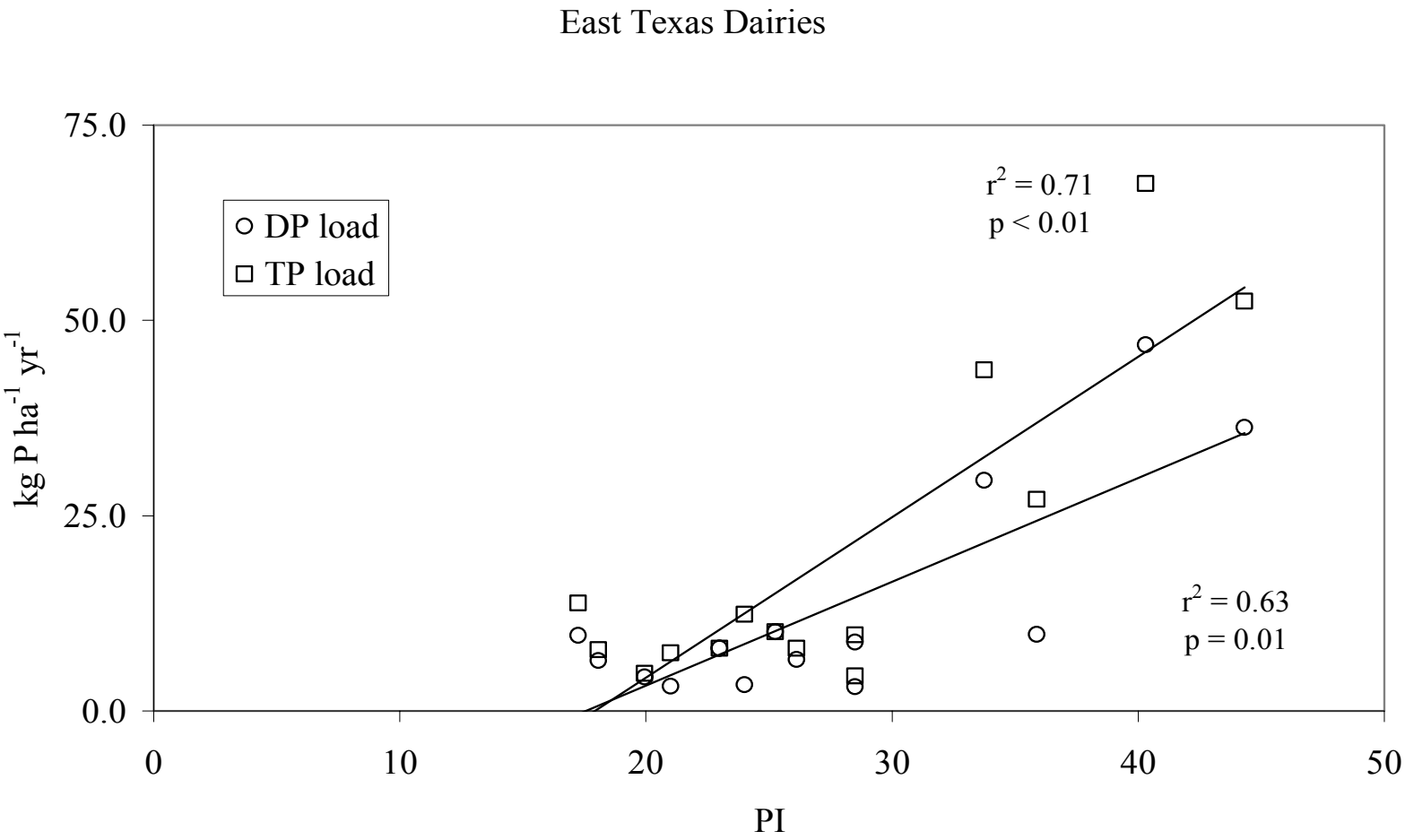

Fig. 26. Relationship between P index (PI) rating (using extractable Mg), dissolved (DP) and total P (TP) loads in runoff on sites with no grazing cattle.

Conclusions

These results indicate that further modifications are needed before the PI rating system is capable of predicting DP and TP losses, at least for the eastern portion of Texas. Part of the problem with the eastern (Central Texas) regions is the wider range in soil pH, which in turn affects the proportion of elements (Al and Fe or Ca) binding P. Use of the RUSLE2 program to estimate erosion rates increased variability in our efforts to relate runoff P losses from our plots to the PI, probably due to the size of our plots (1.5 x 2 m^2). It is likely that our relationship using erosion estimates would have been similar to our measured results if we had measured DP and TP losses at a larger scale, such as watershed or at least per hectare. Our design is also for all rainfall events to be at rates of 7.5 cm hr^{-1} which would tend to produce greater losses, particularly in the High Plain regions. Modification of the PI to include extractable soil Mg improved the overall relationship between P index ranking and P losses in runoff. Studies using poultry manure and commercial inorganic fertilizers are currently underway to examine if extractable Mg can be use to explain dissolve P release. Numerous studies have shown a clear difference in runoff dissolve P concentrations from fields amended at same P rates with different manure sources or inorganic fertilizers (Kleinman and Sharpley, 2003; Siddique and Robinson 2004; Tarkalson and Mikkelsen, 2004). Further experiments to identify whether the effects of Mg on P release are due to a binding mechanism or an indirect effect should be pursued.

While our findings showed many shortcomings in the prediction of both DP and TP losses based on site ranking, a relationship was present and greater than trying to use

soil P test values alone. Overall, the model reflected the intuitive notion that good stands of grass-covered fields have the lowest vulnerability to contribute P by reducing sediment as well as dissolved and total P loads in runoff. Inclusion of DP as a parameter for the current index is not promising as its estimation is not possible under calcareous conditions for different soils. Like any model, we are restricted by the quality of data that we can obtain. Although errors in our experimental protocol were identified and often corrected, we were dependent on the quality of cooperation and the reliability of that data from producers. Efforts to bridge this gap are being attempted to further improve the reliability of the P index. The findings here support others' view that no attempts should be made to use a single test such as a P threshold to regulate P application from an environmental aspect.

CHAPTER VI

SUMMARY

Attempts to minimize the effects of eutrophication on surface water due to P inputs from agricultural nonpoint sources are crucial in environmentally sensitive areas. Two examples of these areas are the Leon and Bosque watersheds in Central Texas, where a large number of dairies are located. In order to address this issue in large livestock operations, new management plans are being created. One new tool being used in Texas is the Phosphorus Index (PI), which attempts to rank sites based on the relative potential to contribute P through runoff. The advantage of this approach is that it utilizes a number of source and transport factors that are site specific. Studies to verify the effectiveness of the PI in Texas included soil P testing reliability, response to application of manure among different soil series, and correlation of site PI rankings to measured P losses in the runoff from various sites throughout Texas.

Soil P testing has been used as a way to estimate crop need and has usually involved colorimetric techniques. Today most labs use ICP techniques for multi-nutrient testing. Testing the reliability of soil test procedures using ICP involved 21 sites across seven states that had various manure sources (poultry, swine, beef, and dairy) applied at different rates for continuous periods up to 50 years. The methods studied included four fertility tests: Olsen, Bray I, Mehlich III, and TAMU, the last two having been used by the Texas Cooperative Extension services as the primary test. Two dilute salt solutions ($CaCl_2$ and KCl), and deionized water. Our studies showed that reproducibility and the order of extracting P efficiency were consistent and independent of manure source

(poultry, swine, dairy, and beef) for ICP techniques of analysis for acidic to slightly neutral soil pH conditions. Mehlich III consistently extracted the most P followed in decreasing order by Bray I, Olsen, TAMU, deionized water, KCl, and $CaCl_2$. As conditions became more calcareous, Bray I P concentrations dropped to those of deionized water while its variability rose to unacceptable levels. Meanwhile the extracting power for TAMU increased gradually until it surpassed those of Mehlich III. This was a result of the longer extracting period used for the TAMU method that allowed for greater dissolution of the less soluble calcium bound phosphorus portions found under calcareous conditions. Determination of Mehlich III P concentration using ICP techniques was shown to be greater than when measurements were performed colorimetrically. The greater concentration using ICP are due to the inclusion of any soluble organic bound P present in solution whereas colorimetric analyses can only measure inorganic P. TAMU P measured colorimetrically was lower when extracting levels were below 350 mg kg-1 but eventually increased and consistently surpassed colorimetric values for the same plot as extractable P concentrations increased.

Surface application of varied amounts of dairy manure led to proportional increases of STP values (Mehlich III, TAMU P, and CSSP) in four benchmark soil series from the Bosque watershed. A highly linear relationship was present for sampling depths ranging from the top 2.5 cm down to 15 cm. In these cases, incubation periods were at least a year and livestock was kept off fields. Dissolved P concentrations in runoff from all soil series were correlated by first order regression lines to the STP concentrations using Mehlich III, TAMU, and CSSP extractions. However no single

linear equation could be use all soils to describe the estimate DP using STP values. The equations of linear regression however were not significantly equal for all soil series when surface soil samples were used. As sampling depth increased, the ability to statistically differentiate among soil series was lost for all STP methods used except the TAMU extraction. Using TAMU P, we could determine differences in the STP and DP relationship among soil series down to a 15 cm sampling depth. The reason for this difference was due to the lower release of dissolved P from highly calcareous soils compared to other soils, even when they had equivalent STP values. This would indicate that at least for high pH soils, the use of a single relationship to predict dissolved P in the runoff is not feasible.

Attempts to show a relationship between STP values and DP concentrations in runoff from fields in CAFOs with various soil series, application rates, time of incubation of manure which affects mineralization and fixation rates, and management plans yielded mixed results. Under acid to slightly neutral soil conditions ($pH < 6.4$) we were able to establish significant correlations for all extracting methods and sampling depths down to 15 cm for four different soil series. Indicating that under these conditions pH is a more important factor than soil series properties for the estimations of DP into runoff based on STP. This is an indication that most of the P found is bound by relatively soluble Al and Fe oxides. In high soil pH plots, indicative of high $CaCO_3$ content, determination of DP release was not significant. This agrees with preliminary studies in fenced plots with high soil pH, where various manure rates were surface applied and showed that the relationship to be soil type specific. All indications are that

estimation of DP release by soil P testing will not be possible, at least not under calcareous conditions. However, extracted soil Mg with the TAMU extractant proved to be a relative good indicator of not only DP and TP concentrations in runoff, but of field loads for both parameters. This was attributed to likelihood that Mg bound phosphate is a major form of compounds found in lagoons where effluents are stored prior to application. More research is needed to verify if Mg extracted by other methods can be equally related to DP and TP released in runoff, if so this category could be incorporated into the current PI rating system.

By far the greater effect on P losses from fields was found to be the type of vegetative cover and land cultivation practice at time of rainfall simulation. Grass cover promoted infiltration as well as intercepted rainfall to such levels that it required the greatest amounts of simulated rain before any runoff could be generated regardless of field slope. Although the percentage of DP loss was the highest for grass-covered fields, the overall decrease in volume as well as DP and TP concentrations with regard to freshly tilled soils, those with row crops or row crop residues, resulted in the lowest overall losses. While the concentrations of nutrients decreased over time, the percent DP remained constant throughout the first 30 minutes of a rainfall simulation, with final concentrations of P in the runoff determined by the amount released during the first few minutes (< 15 min.) of the runoff event.

Total soil P content influenced runoff DP and TP concentrations, but only when the content was below 700 mg kg^{-1}, after which point all correlations degenerated. It is

likely that this is due to bound P precipitating rather than being adsorbed by soil particles.

Finally, through the use of twenty-three sites throughout Texas we were able to show that, when sites were averaged by vegetation type and management practices, the PI rating order coincided with actual losses. However this relationship was poor when all individual sites were plotted for the East Texas PI. Modification of the PI by inclusion of a soil extractable Mg category greatly increased correlations. Further alterations should probably include a revision of STP ranking since practically every field that has had effluent or solid manure applied will exceed crop recommendations. These results illustrate the limitation of this system to quantify P losses based on so many different factors (transportation and sources) and their interactions. It is currently not feasible or practical to use universal regulations for manure application based on phosphorus content of soils or manure. Although initial assumptions were relatively good, further regional and geological modifications are still needed.

REFERENCES

Andraski, B.J., D.H. Mueller, and T.C. Daniel. 1985. Phosphorus losses in runoff as affected by tillage. Soil Sci. Soc. Am. J. 49:1523-1527.

Ashworth J., and K. Mrazek. 1995. Modified Kelowna test for available phosphorus and potassium in soil. Commun. Soil Sci. Plant Anal. 26:731-739.

Baker, R.T., and J.K. Hall. 1967. Measurements of phosphorus availability in acid soils of Pennsylvania. Soil Sci. Soc. Am. Proc. 31:662-667.

Booram, C.V., T.E. Hazen, and R.J. Smith. 1975. Trends and variations in an anaerobic lagoon with recycling. p. 537-540. *In* Managing livestock waste. Proc. Int. Symp. on Livestock Wastes. Am. Soc. Agric. Eng., St Joseph, MI.

Bowman, R.A. and C.V. Cole. 1978. An exploratory method for fractionation of organic phosphorus from grassland soils. Soil Sci. 125:95-101.

Boyce, F.M., M.N. Charlton, D. Rathke, C.H. Mortimer, and J. Bennett. 1987. Lake Erie research: Recent results, remaining gaps. J. Great Lakes Res. 13:826-840.

Brady, N.C. 1990. Nature and properties of soils. Macmillan Publishing Company, New York.

Brady, N.C and R.R. Weil. 1999. The nature and properties of soils. Prentice-Hall, Inc., Upper Saddle River, NJ.

Bray, R.H., and L.T Kurtz, 1945. Determination of total, organic, and available forms of phosphorus in soils. Soil Sci. 59:39-45.

Cole, C.V., S.R. Olsen, and C.O. Scott. 1953. The nature of phosphate sorption by calcium carbonate. Soil Sci. Soc. Am. Proc. 17:352-356.

Correll, D.L. 1981. Eutrophication trends in the water quality of the Rhode River (1971-1978). p. 425-435. *In* B.J. Neilson and L.E. Cronin (ed.). Estuaries and nutrients. Humana Press, Clifton, NJ.

Correll, D.L. 1998. The role of phosphorus in the eutrophication of receiving waters: A review. J. Environ. Qual. 27:261-266.

Dalal, R.C., and E.G. Hallsworth. 1977. Measurement of isotopic exchangeable soil phosphorus and interrelationship among parameters of quantity, intensity, and capacity factors. Soil Sci. Soc. Am. J. 41:81-86.

Daniel, T.C. 1999. Rainfall Simulator Workshop: Initial plot wet up, antecedent moisture determinations, multiple simulation runs, and field moisture capacity. University of Arkansas, Fayetteville, AK.

Daverede, I.C., A.N. Kravchenko, R.G. Hoeft, E.D. Nafziger, D.G. Bullock, J.J. Warren, and L.C. Gonzini. 2004. Phosphorus runoff from incorporated and surface-applied liquid swine manure and phosphorus fertilizer. J. Environ. Qual. 33:1535-1544.

Destain, J.P. 1980. The effect of very intensive dressings of livestock effluent on phosphorus in soil. p. 371-381. *In* Hucker, T.W.G., and C. Catroux (eds). Phosphorus in sewage sludge and animal waste slurries: Proceedings of the EEC Seminar organized jointly by the CEC and the Institute for Soil Fertility. D. Reidel Publishing Co., London, England.

Dick, W.A., and M.A. Tabatabai. 1977. Determination of orthophosphate in aqueous solutions containing labile organic and inorganic phosphorus compounds. J. Environ. Qual. 6:82-85.

Dremanis, A. 1962. Quantitative gasometric determination of calcite and dolomite by using the Chittick apparatus. J. Sedimentary Petrology. 32:520-529.

Eck, H.V., and B.A. Stewart. 1995. Manure. p. 169-198. *In* J.E. Rechcigl (ed.) Soil amendments and environmental quality. Lewis Publishers, Boca Raton, FL.

Edwards, D.R., and T.C. Daniel. 1993. Effects of poultry litter application rate and rainfall intensity on quality of runoff from fescue grass plots. J. Environ. Qual. 22:361-365.

Feagley, S.E., M. Dollar, J. Childers, J. Crownover, D. Embry, B. Hanna, and R. Leopold. 2004. Chapter III: Animal manures and biosolids; Chapter IV: Nutrient management. *In* NRCS/TAEX nutrient management specialist certification course, Texas A&M University.

Feagley, S.E., M.S. Valdez, and W.H. Hudnall. 1994. Bleached, primary papermill sludge effect on bermudagrass grown on a mine soil. Soil Science 157(6)389-397.

Ferguson, J. F., D. Jenkins, and J. Eastman. 1973. Calcium phosphate precipitation at slightly alkaline pH values. J. Water Pollution Control Fed. 45:620-631.

Fox, R.L. and E.J. Kamprath. 1971. Adsorption and leaching of P in acid organic soils and high organic matter sand. Soil Sci. Soc. Am. Proc. 35:154-156.

Furrer, O.J. 1980. Accumulation and leaching of phosphorus as influenced by sludge application. p. 235-240. *In* Hucker, T.W.G., and C. Catroux (eds). Phosphorus in sewage sludge and animal waste slurries: Proceedings of the EEC Seminar organized jointly by the CEC and the Institute for Soil Fertility. D. Reidel Publishing Co., London, England.

Gartley, K.L. and J.T. Sims. 1994. Phosphorus soil testing: Environmental uses and implications. Commun. Soil Sci. Plant Anal. 25:1565-1582.

Gburek, W.J., A.N. Sharpley, L. Heathwaite, and G.J. Folmar. 2000. Phosphorus management at the watershed scale: A modification of the phosphorus index. J. Environ. Qual. 29:130-144.

Gerritse, R.G. 1980. Mobility of phosphorus from pig slurry in soils. p. 347-369. *In* Hucker, T.W.G., and C. Catroux (eds). Phosphorus in sewage sludge and animal waste slurries: Proceedings of the EEC Seminar organized jointly by the CEC and the Institute for Soil Fertility. D. Reidel Publishing Co., London, England.

Grossl, P.R. and W.P. Inskeep. 1991. Precipitation of dicalcium phosphate dihydrate in the presence of organic acids. Soil Sci. Soc. Am. J. 55:670-675.

Hansen, J.C., B.J. Cade-Menum, and D.G. Strawn. 2004. Phosphorus speciation in manure-amended alkaline soils. J. Environ. Qual. 33:1521-1527.

Harris, W.G., R.D. Rhue, G. Kidder, R.B. Brown, and R. Littell. 1996. Phosphorus retention as related to morphology of sandy coastal plain soil materials. Soil Sci. Am. J. 60:1513-1521.

Harris, W.G., H.D. Wang, and K.R. Reddy. 1994. Dairy manure influence on soil and sediment composition: Implications for P retention. J. Environ. Qual. 23:1071-1081.

Harrison, A.G. 1987. Soil organic phosphorus. A review of world literature. CAB International, Wallingford, UK.

He, Z., T.S. Griffin, and C.W. Honeycutt. 2004. Phosphorus distribution in dairy manures. J. Environ. Qual. 33:1528-1534.

Hedley, M.J., J.W.B. Stewart, and B.S. Chauhan. 1982. Changes in inorganic and organic soil phosphorus fractions induced by cultivation practices and by laboratory incubations. Soil Sci. Soc. Am. J., 46:970-976.

Holford, I.C.R., and G.E.G Mattingly. 1975. The high and low-energy phosphate absorbing surfaces in calcareous soils. J. Soil Sci. 26:407-417.

Hons, F.M., L.A. Larson-Vollmer, and M.A. Locke. 1990. NH_4OAc-EDTA-extractable phosphorus as a soil test procedure. Soil Sci. 149:249-256.

Indiati, R., and A.N. Sharpley. 1997. Changes in some soil phosphorus availability parameters as induced by phosphorus addition and soil sorption properties. Commun. Soil Sci. Plant Anal. 29:1565-1578.

Inskeep, W.P., and J.C. Silvertooth. 1988. Inhibition of hydroxyapatite precipitation in the presence of fulvic, humic and tannic acids. Soil Sci. Soc. Am. J. 52:941-946.

Jackson, T.L., J. Hay, and D.P. Moore. 1967. The effect of Zn on yield and chemical compositon of sweet corn in the Willamette Valley. Proc. Am. Soc. Hortic. Sci. 91:462-471.

Johnston, A.E. 1980. Accumulation of phosphorus in a sandy soil from farmyard manure (FYM) and sewage sludge. p. 273-290. *In* Hucker, T.W.G., and C. Catroux (eds). Phosphorus in sewage sludge and animal waste slurries: Proceedings of the EEC Seminar organized jointly by the CEC and the Institute for Soil Fertility. D. Reidel Publishing Co., London, England.

Jones, J.B. 1998. Soil test method: Past, present, and future use of soil extractants. Commun. Soil Sci. Plant Anal. 29:1543-1552.

Kleinman, P.J.A., and A. Sharpley. 2003. Effect of broadcast manure on runoff phosphorus concentrations over successive rainfall events. J. Environ. Qual. 32:1072-1081.

Kleinman, P.J.A., A. Sharpley, T.L. Veith, R.O. Maguire, and P.A. Vadas. 2004. Evaluation of phosphorus transport in surface runoff from packed soil boxes. J. Environ. Qual. 33:1413-1423.

Kuo, S. 1996. Phosphorus. p. 869-919. *In* D.L. Sparks et al. (ed.) Methods of soil Analysis Part 3. Chemical Method No.5. Soil Science Society of America: American Society of Agronomy. Madison, WI.

Kuo, S., and E.J. Jellum. 1987. Influence of soil characteristics and environmental conditions on seasonal variation of water-soluble phosphate in soils. Soil Sci. 143:257-263.

Labhsetwar, V.K., and P.N. Soltanpour. 1985. A comparison of NH_4HCO_3-EDTA, $NaHCO_3$, $CaCl_2$, and Na_2-EDTA soil tests for phosphorus. Soil Sci. Soc. Am. J., 49:1437-1440.

Lander, C.H., D. Moffitt, and K. Alt. 1998. Nutrients available from livestock manure relative to crop growth requirements. Resource Assessment and Strategic Planning Working Paper 98-1 [Online]. Available at http://www.nhq.nrcs.usda.gov/land/pubs/nlweb.html (verified 5 Oct. 2004).

Lanynyon, L.E. and PB. Thompson. 1996. Changing emphasis of farm production. p. 15-23. *In* M. Salis and J. Popow (eds.) Animal agriculture and the environment: Nutrients, pathogens, and community relations. Northeast Regional Agricultual Engineering Services, Ithaca, NY.

Larsen, K.E. 1980. Phosphorus effect of animal manure and sewage sludge. p.207-228. *In* Hucker, T.W.G., and C. Catroux (eds). Phosphorus in sewage sludge and animal waste slurries: Proceedings of the EEC Seminar organized jointly by the CEC and the Institute for Soil Fertility. D. Reidel Publishing Co., London, England.

Lemunyon, J.I., and R.G. Gilbert. 1993. The concept and need for a P assessment tool. J. Prod. Agric. 6:483-486.

Maloney, T.E., W.E. Miller, and T. Shiroyama. 1972. Algal responses to nutrient additions in natural water. I. Laboratory essays. p.134-140. *In* G.E. Likens (ed.). Nutrients and eutrophication. Spec. Symp. Vol. 1. Am. Soc. Limnol. Oceanogr. Allen Press, Lawrence, KS.

Marschner, H. 1995. Mineral nutrition of higher plants 2nd Ed. p. 265-277. Academic Press, San Diego, CA.

McDowell, R.W., and A.N. Sharpley. 2001. Approximating phosphorus release from soils to surface runoff and subsurface drainage. J. Environ. Qual. 30:508-520.

McFarland, A.M.S., and L.M. Hauck. 1999. Relating agricultural land uses to in-stream stormwater quality. J. Environ. Qual. 28:836-844.

Mehlich, A. 1984. Mehlich 3 soil test extractant: A modification of Mehlich 2 extractant. Commun. Soil Sci. Plant Anal., 15:1409-1416.

Mehta, N.C., J.O. Legg, C.A.I. Goring, and C.A. Black. 1954. Determination of organic phosphorus in soils: 1. Extraction methods. Soil Sci. Soc. Am. Proc. 13:443-449.

Miller, W.P. 1987. A solenoid-operated, variable intensity rainfall simulator. Soil Sci. Am. J. 51:832-834.

Morse, D., H.H. Head, and C.J. Wilcox. 1992. Disappearance of phosphorus in phytate from concentrates in vitro and from rations fed to lactating cows. J. Dairy Sci. 75:1979-1986.

Murphy, J., and J.P. Riley. 1962. A modified single solution method for the determination of phosphate in natural waters. Anal. Chim. Acta 27:31-36.

National Research Council. 1993. Soil water quality: an agenda for agriculture. National Academy Press, Washington, DC.

O'Connor, G.A., K.L. Knudtsen, and G.A. Connel. 1986. Phosphorus solubility in sludge-amended calcareous soils. J. Environ. Qual. 15:308-312.

Olsen S.R., C.V. Cole, F.S. Watanabe, and L.A. Dean. 1954. Estimation of available phosphorus in soils by extraction with sodium bicarbonate. USDA Circ. 9939. USDA, Washington, DC.

Olsen, S.R., and L.E. Sommers. 1982. Phosphorus. p. 403-430. *In* A.L. Page et al. (ed). Methods of soil analysis. Part 2. 2nd ed. Agron. Monogr. 9. ASA and SSSA. Madison, WI.

Olsen S.R., and F.S. Watanabe. 1970. Diffusive supply of phosphorus in relation to soil textural variations. Soil Sci. 110:318-327.

Parry, R. 1998. Agricultural phosphorus and water quality: a U.S. environmental protection agency perspective. J. Environ. Qual. 27:258-261

Penn, C.J., G.L. Mullins, L.W. Zelazny, J.G. Warren, and J.M. McGrath. 2004. Surface runoff losses of phosphorus from Virginia soils amended with turkey manure using phytase and high available phosphorus corn diets. J. Environ. Qual. 33:1431-1439.

Peters, R.H. 1981. Phosphorus availability in Lake Memphremagog and its tributaries. Limnol. Oceanogr. 26:1150-1161.

Pierzynski, G.M., J.T. Sims, and G.F. Vance. 2000. Soils and environmental quality. 2nd Ed. Lewis Publishers, Boca Raton, FL.

Pote, D.H., and T.C. Daniel. 2000. Analyzing for total phosphorus and total dissolved phosphorus in water samples. p. 97-97. *In* Pyerzynski, G.M.(ed.) Methods of phosphorus analysis: for soils, sediments, residuals, and waters. Southern Cooperative Series Bulletin #396. SERA-IEG 17 regional publication. [Online]. Available at http://www.soil.ncsu.edu/sera17/publications/era17-2/pm_cover.htm (verified 10 Oct. 2004).

Pote, D.H., T.C. Daniel, D.J. Nichols, A.N. Sharpley, P.A. Moore, Jr., D.M. Miller, and D.R. Edward. 1999. Relationship between phosphorus levels in three Ultisols and phosphorus concentrations in runoff. J Environ. Qual. 28:170-175.

Pote, D.H., T.C. Daniel, A.N. Sharpley, P.A. Moore, Jr., D.R. Edwards, and D.J. Nichols. 1996. Relating extractable soil phosphorus to phosphorus losses in runoff. Soil Sci. Soc. Am J. 60:855-859.

Powers, C.F., D.W. Schults, K.W. Malueg, R.M. Brice, and M.D. Schuldt. 1972. Algal responses to nutrient addition in natural waters. II. Field experiments. p. 141-156. *In* G.E. Likens (ed.) Nutrients and eutrophication. Spec. Symp. Vol. 1. Am. Soc. Limnol. Oceanogr., Allen Press, Lawrence, KS.

Provin, T. 2003. Texas Cooperative Extension Soil, Water, and Forage Testing Laboratory standard operating procedures. College Station, TX.

Rechcigl, J.E.(ed.) 1995. Soil amendments and environmental quality. Lewis Publishers, Boca Raton, FL.

Redfield, A.C. 1958. The biological control of chemical factors in the environment. Am. Sci. 46:1-221.

Rowell, D.L. 1994. Soil science: Methods and application. p. 205. Longman Scientific & Technical. Harlow, England.

RUSLE2. 2004. Revised universal soil loss equation version 2. [Online]. Available at http://fargo.nserl.purdue.edu/rusle2_dataweb/RUSLE2_Index.htm (verified 10 Oct. 2004).

Saunders, W.M., and E.G. Williams. 1955. Observations on the determination of total organic phosphorus in soils. J. Soil Sci. 6:254-267.

Schelske, C.L., and E.F. Stoemer. 1972. Phosphorus, silica, and eutrophication of Lake Michigan. p. 157-171. *In* G.E. Likens (ed.) Nutrients and eutrophication. Spec. Symp. Vol. 1. Am. Soc. Limnol. Oceanogr., Allen Press, Lawrence, KS.

Schindler, D.W. 1977. The evolution of phosphorus limitation in lakes. Science (Washington, DC) 195:260-262.

Schroeder, P.D., D.E. Radcliffe, M.L. Cabrera, and C.D. Belew. 2004. Relationship between soil test phosphorus and phosphorus in runoff: Effects of soil series variability. J. Environ. Qual. 33:1452-1463.

Sharpley, A.N. 1993. An innovative approach to estimate bioavailable phosphorus in agricultural runoff using iron oxide-impregnated paper. J. Environ. Qual. 22:597-601.

Sharpley, A.N. 1995a. Dependence of runoff phosphorus on extractable soil phosphorus. J. Environ. Qual. 24:920-926.

Sharpley, A.N. 1995b. Identifying sites vulnerable to phosphorus loss in agricultural runoff. J. Environ. Qual. 24:947-951.

Sharpley, A.N., S.C. Chapra, R. Wedepohl, J.T. Sims, T.C. Daniel, and K.R. Reddy. 1994. Managing agricultural phosphorus for protection of surface waters: Issues and options. J. Environ. Qual. 23:437-451.

Sharpley, A.N., B. Foy, and P. Withers. 2000. Practical and innovative measures for the control of agricultural phosphorus losses to water: An overview. J. Environ. Qual. 29:1-9.

Sharpley, A.N., P. Kleinman. 2003. Effect of rainfall simulator and plot scale on overland flow and phosphorus transport. J. Environ. Qual 32:2172-2179.

Sharpley, A.N., and B. Moyer. 2000. Phosphorus forms in manure and compost and their release during simulated rainfall. J. Environ. Qual. 29:1462-1469.

Sharpley, A.N., and S.J. Smith. 1989. Mineralization of and leaching of phosphorus from soil incubated with surface-applied and incorporated crop residue. J. Environ. Qual. 18:101-105.

Sharpley, A.N., S.J. Smith, O.R. Jones, W.A. Berg, and G.A. Coleman. 1992. The transport of bioavailable phosphorus in agricultural runoff. J. Environ. Qual. 21:30-35.

Sharpley, A.N., S.J. Smith, B.A. Stewart, and A.C. Mathers. 1984. Forms of phosphorus in soil receiving cattle feedlot waste. J. Environ. Qual. 13:211-215.

Shelton, C.H., R.D. von Bernuth, S.P. Rajbhandari. 1985. A continuous-application rainfall simulator. ASAE 28(4):1115-1119

Shreve, B.R., P.A. Moore, T.C. Daniel, D.R. Edward, and D.M. Miller. 1995. Reduction of phosphorus in runoff from field-applied poultry litter using chemical amendments. J. Environ. Qual. 24:106-111.

Siddique, M.T., and J.S. Robinson. 2004. Differences in phosphorus retention and release in soils amended with animal manures and sewage sludge. Soil Sci. Soc. Am. J. 68:1421-1428.

Simons, P.C.M., and A.W. Jongbloed. 1980. Influence of nutritional and physiological factors on phosphorus content of animal manures. p. 119-133. *In* Hucker, T.W.G., and C. Catroux (eds). Phosphorus in sewage sludge and animal waste slurries: Proceedings of the EEC Seminar organized jointly by the CEC and the Institute for Soil Fertility. D. Reidel Publishing Co., London, England.

Sims, J.T. 1992. Environmental management of phosphorus in agricultural and municipal wastes. p. 59-65. *In* Future directions for agricultural phosphorus research, Bulletin Y-224, National Fertilizer and Environmental Research Center, Muscle Shoals, AL,

Sims, J.T. 1993. Environmental soil testing for phosphorus. J. Prod. Agric. 6:501-506.

Sims, J.T., R.R. Simard, and B.C. Joern. 1998. Phosphorus loss in agricultural drainage: Historical perspective and current research. J. Environ. Qual. 27:277-293.

Smith, R.A., R.B. Alexander, and M.G. Wolman. 1987. Water-quality trends in the nation's rivers. Science (Washington, DC) 235:1607-1615.

Soltanpour P.N., F. Adams, and A.C. Bennett. 1974. Soil phosphorus availability as measured by displaced soil solutions, calcium-chloride extracts, dilute-acid extracts, and labile phosphorus. Soil Sci. Soc. Am. Proc. 38:225-228.

Soltanpour, P.N., R.L. Fox, and R.C. Jones. 1988. A quick method to extract organic phosphorus from soils. Soil Sci. Soc. Am. J. 51:255-256.

Sonzongoni G., D.R. Chesters, D.N. Jeff, J.C. Konrad, R.C. Ostry, and J.B. Robinson. 1980. Pollution from land runoff. Environ. Sci. Tech. 14:148-153.

SPSS. 2001. SPSS Version 11.0.1. SPSS, Chicago, IL.

Stevenson, F.J. 1999. Cycles of soil, carbon, nitrogen, phosphorus, sulfur, micronutrients. John Wiley and Sons, Inc., New York.

Suss, A. 1980. Phosphorus problems and sewage sludge application. p. 191-205. *In* Hucker, T.W.G., and C. Catroux (eds). Phosphorus in sewage sludge and animal waste slurries: Proceedings of the EEC Seminar organized jointly by the CEC and the Institute for Soil Fertility. D. Reidel Publishing Co., London, England.

Tan, K.H. 1996. Soil sampling, preparation and analysis: Determination of available P. p. 157-167. Marcel Dekker Inc. New York.

Tarkalson, D.D. and R.L. Mikkelsen. 2004. Runoff phosphorus source, application method, and application rate on a Piedmont soil. J. Environ. Qual. 33:1424-1430.

Texas A&M University Agricultural Extension Service 1980. Soil testing procedures. College Station, TX.

Tiessen, H., J.W.B. Stewart, and J. O. Moir. 1983. Changes in organic and inorganic phosphorus composition of two grassland soils and their particle size fractions during 60-90 years of cultivation. J. Soil Sci. 34:815-823.

Torbert, H.A., T.C. Daniel, J.L. Lemunyon, and R.M. Jones. 2002. Relationship of soil test phosphorus and sampling depth to runoff phosphorus in calcareous and noncalcareous soils. J. Environ. Qual. 31:1380-1387.

Turner, B.L., M.A. Kay, and D.T. Westerman. 2004. Phosphorus in surface runoff from calcareous arable soils of the semiarid western United States. J. Environ. Qual. 33:1814-1821.

Udawatta, R.P., P.P. Motavalli, and H.E. Garrett. 2004. Phosphorus loss and runoff characteristics in three adjacent agricultural watersheds with claypan soils. J. Environ. Qual. 33:1709-1719.

U.S. Department of Agriculture/Natural Resources Conservation Service. 2000a. Conservation Practice Standard Nutrient Management Code 590. [Online]. Available at http://www.nrcs.usda.gov/technical/ECS/nutrient/590.html (verified 8 Jan. 2005)

U.S. Department of Agriculture/Natural Resources Conservation Service. 2000b. Phosphorus assessment tool for Texas. Agronomy Tech. Note Number 15. [Online]. Available at http://www.pswmru.arsup.psu.edu/phosphorus/TexasPI (verified 5 Oct. 2004).

U.S. Environmental Protection Service (EPA). 1986. Quality criteria for water. USEPA Rep. 440/5-86-001. USEPA, Office of Water Regulations and Standards. U.S. Gov. Print. Office (PB87-226759), Washington, DC. [Online]. Available at http://www.epa.gov/waterscience/criteria/goldbook.pdf

U.S. Environmental Protection Service (EPA). 1996. Environmental indicators of water quality in the United States. EPA 841-R-96-002. Washington, DC.

U.S. Environmental Protection Service (EPA). 2000. Nutrient criteria technical guidance manual for lakes and reservoirs. EPA 822-B00-001. Washington, DC.

United States Geological Survey. 1999. The quality of our nation's waters: Nutrients and pesticides. USGS Circ. 1225. U.S. Gov. Print. Office, Washington, DC.

Vadas, P.A., J.J. Meisinger, L.J. Sikora, J.P. McMurtry, and A.E. Sefton. 2004. Effect of poultry diet on phosphorus in runoff from soils amended with poultry manure and compost. J. Environ. Qual. 33:1845-1854.

Van Diest, A. 1963. Soil test correlation studies on New Jersey soils: Comparisons of seven methods for measuring labile inorganic soil phosphorus. Soil Sci. 96:261-266.

Van der Paauw, F. 1971. An effective water extraction method for the determination of plant available soil phosphorus. Plant Soil 34:467-481.

Vetter, H., and G. Steffens. 1980. Phosphorus accumulation in soil profiles and phosphorus losses after the application of animal manures. p. 309-331. *In* Hucker, T.W.G., and C. Catroux (eds). Phosphorus in sewage sludge and animal waste slurries: Proceedings of the EEC Seminar organized jointly by the CEC and the Institute for Soil Fertility. D. Reidel Publishing Co., London, England.

Webber, M.D. 1980. Phosphorus in relation to water quality: Great Lakes water quality and the phosphorus removal program. p. 5-19. *In* Hucker, T.W.G., and C. Catroux (Eds). Phosphorus in sewage sludge and animal waste slurries: Proceedings of the EEC Seminar organized jointly by the CEC and the institute for soil fertility. D. Reidel Publishing Co., London, England.

Wen, G., T.E. Bates, R.P. Voroney, J.P. Winter, and M.P. Schellenbert. 1997. Comparison of phosphorus availability with application of sewage sludge, sludge compost and manure compost. Commun. Soil Sci. Plant Anal. 28:1481-1497.

Withers, J.W., and A.N. Sharpley. 1995. Phosphorus fertilizers. p. 65-107. *In* J.E. Rechcigl (ed.) Soil amendments and environmental quality. Lewis Publishers, Boca Raton, FL.

White R.E., and P.H.T. Beckett. 1964. Studies on the phosphate potentials of soils. I. The measurement of phosphate potential. Plant Soil 20:1-16.

Williams, J.D.H., J.K. Syers, T.W. Walker, and R.W. Rex. 1970. A comparison of methods for the determination of soil organic phosphorus. Soil Sci. 110:13-18.

Wood, C.W. 1998. Agricultural phosphorus and water quality: an overview. p. 5-12. *In* Sims J.T. (ed.) Soil testing for phosphorus – Environmental uses and implications. Southern Cooperative Series Bulletin No.389 [Online]. Available at http://www.soil.ncsu.edu/sera17/publications/sera17-1/Intro.htm#IndexChap1 (verified 8 Jan. 2005).

Yee, A.R., and K. Broersma. 1987. The Bray, Mehlich and Kelowna soil P tests as affected by soil carbonates. Can. J. Soil Sci. 67:399-404.

Zech, W., H.G. Alt, L. Haumair, and R. Blasek. 1987. Characterization of P fractions in mountain soils of Bavarian Alps by ^{32}P-NMR spectroscopy. p. 559. *In* Marschner H., 1995. Mineral nutrition of higher plants, 2nd Ed. Academic Press, San Diego, CA.

APPENDIX

Table A-1. Level of significance (p) averaged among all soil series.

Extraction	Sampling Depth 0- 2.5 cm	0- 5 cm	0- 15cm	all depths
	p	p	p	p
SSSP	0.03	0.11	0.17	0.11
Mehlich III	0.04	0.03	0.15	0.07
TAMU	0.05	0.02	0.05	0.04

Table A-2. Runoff conductivity and pH changes due to time.

Plot	Slope	15 min.	30 min.	composite	15 min.	30 min.	composite
		conductivity(μS)			pH		
A	1.2	†	†	†	†	†	†
B	5.8	†	†	†	†	†	†
C	0.2	†	†	†	†	†	†
D	0.3	70.0	38.5	61.4	8.97	8.81	8.72
E	0.2	†	†	127.6	†	†	7.51
F	4.5	114.3	82.0	130.3	7.75	8.27	8.08
G	0.8	90.4	70.6	101.8	†	†	7.80
H	4.4	141.6	109.0	131.1	7.31	7.43	7.39
I	3.9	129.2	101.1	133.9	7.52	7.54	7.53
J	4.2	148.2	79.9	148.3	6.46	6.31	6.41
K	3.7	257.7	176.7	285.7	7.12	6.97	7.21
L	2.2	130.0	89.7	150.7	8.97	9.14	8.96
M	7.6	72.8	55.1	79.1	7.34	7.46	7.46
N	6.7	82.3	67.0	86.2	7.68	7.72	7.69
O	5.9	82.8	65.3	83.6	7.21	7.17	7.29
P	8.1	77.3	58.8	87.3	7.10	7.12	7.19
Q	10.3	37.4	28.5	45.5	7.25	7.23	7.38
R	4.5	37.7	32.3	39.8	7.49	7.43	7.55
S	2.1	48.0	36.8	48.7	7.50	7.52	7.54
T	3.0	30.2	28.4	32.2	7.55	7.64	7.64
U	1.9	11.9	9.3	15.6	7.10	7.03	7.21
V	4.3	95.5	87.6	99.3	9.65	9.67	9.65
W	2.5	95.9	70.0	102.7	9.35	9.47	9.39

† Not measured

VITA

Freddy J. Jacoby
3853 Whistle Train Rd.
Brea, CA 92823

Education

California State University, Los Angeles
Bachelor of Science, Biochemistry Dec. 1992

Clemson University
Master of Science, Agronomy Dec. 1997

Texas A&M University
Ph.D, Soil Science May 2005

Publications

1. Effects of diethylpyrocarbonate on the allosteric properties of phosphoenolpyruvate carboxylase from Crassula argenta. 1991 Taghizadeh SK, Jacoby FJ, Grover SD. Plant Physiol 95:1237-1242

2. Interaction of PEP carboxylase with the photoaffinity label 8-azido AMP. 1991 Jacoby FJ, Newmark N, Grover SD. Plant Physiol 96:85 545

3. Inhibition of DNA repair with aphidicolin enhances sensitivity of targets to tumor necrosis factor. 1993 Gera JF, Fady C, Gardner A, Briskin KB, Jacoby FJ, Lichtenstein A. J. Immunol. 151:3746-3757